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July 22-24, 1970

Sponsored by

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UNIVERSITY OF ALASKA

and

FEDERAL WATER QUALITY ADMINISTRATION

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INTRODUCTION

The first plans for this symposium were begun by the senior editor during the winter of 1967. Alaska was about to become the focal point for petroleum interests as a consequence of the enormous oil discoveries made by the Atlantic-Richfield and Humble Oil joint venture at Prudhoe Bay. Coincident with an economic boom in this sparsely-settled region, federal, state, and private organizations instituted more stringent water pollution control criteria. The combined events created an apparent technology gap relative to water pollution control in cold climates. The engineering and scientific bases for waste treatment and receiving stream criteria were almost exclusively dependent upon temperate experiences. Scattered journal articles about the North were in evidence, but the majority seemed repetitious and contributed little toward a true understanding of the problems involved.

It became obvious that more experience in northern applications was available in the circumpolar countries than in the temperate areas of the United States. A symposium of international scope was therefore envisioned, with representatives of all the circumpolar nations presenting papers. With the exception of the Soviet Union and Iceland, all countries participated. The eighteen papers and two reviews in this volume were selected from approximately sixty invited abstracts.

Persons living or interested in the Far North need no statistics to convince them of the area's potential. It is not the purpose of this introduction to expound on the virtues of the North; however, it is our firm belief that the Arctic and sub-Arctic will be focal points for economic and resource development, and for available open space for the rest of the world in the imminent future. Some implications of this have already become manifest.

Should such development indeed occur, it would certainly be imperative that workers in the circumpolar countries become entirely familiar with the work done in each.

Therefore, besides descriptions of engineered works and scientific studies, the meeting permitted many the opportunity to personally meet their international counterparts. Since such contact is often more important in the long run than the formal papers, hopefully a continuing exchange of information between the various groups concerned with water pollution control will evolve.

The first meeting of such a group is usually difficult. A wide range of disciplines and organizations were involved. The most advanced waste treatment technology is of no use unless the limits of materials able to be discharged into the receiving waters are understood; consequently, invited papers were evenly split between "waste treatment technology" and "effects of wastes upon Far Northern receiving waters." Thus, an attempt was made to create an atmosphere where engineers and biologists could communicate. In this endeavor partial success was achieved. If subsequent meetings are held in the future, it is anticipated that they will be devoted to single, more specific topics than was our general convocation. Indications are that such groups will be meeting in Alaska, Canada, and Scandanavia in the very near future.

The success of any meeting depends upon a large number of people. We thank the authors for their thoughtful manuscript preparation. The secretarial staff and graduate research assistants of the University of Alaska's Institute of Water Resources unselfishly devoted many hours handling the thankless details both in preparation for, and during, the symposium. The banquet talk, which is not included in these papers, was given by Mr. Geoffrey Larminie, area manager for British

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Petroleum (BP Alaska, Inc.). His talk about the Prudhoe Bay oil fields, how they will be and are being managed, and the implications concerning water pollution control will be remembered by all in attendance.

No meeting is successful without financial backing. The majority of funds for the symposium were provided by the Federal Water Quality Administration, U. S. Department of the Interior (now the Environmental Protection Agency), through grant number 16100 E X H. We are particularly indebted to Mr. Richard Latimer, Director of FWQA's Alaska Water Laboratory, and Mr. William Cawley of FWQA's Washington, D. C. staff, for their efforts in making the meeting a success. The Office of Water Resources Research, U. S. Department of the Interior, and the State of Alaska, through their funding of our Institute, contributed significantly, though indirectly, to this meeting. Without their support the symposium could not have been attempted.

The editors sincerely hope this volume will help to bring together some of the existing practices and theories concerned with water pollution control in cold climates.

R. Sage Murphy

David Nyquist

ACKNOWLEDGMENT

In appreciation of financial contributions without which the Symposium could not have been convened.

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TABLE OF CONTENTS

RECEIVING WATERS

SYNOPTIC STUDY OF ACCELERATED EUTROPHICATION IN LAKE TAHOE - AN ALPINE LAKE 1

Charles R. Goldman, Professor of Zoology, Institute of Ecology, University of California, Davis, California, U.S.A.

Gerald Moshiri, Institute of Ecology, University of California, Davis, California, U.S.A.

Evelyne de Amezaga, Institute of Ecology, University of California, Davis, California, U.S.A.

THE SOUTH BASIN OF LAKE WINNIPEG - AN ASSESSMENT OF POLLUTION 22

Jo-Anne M. E. Crowe, Pollution Biologist, Fisheries Branch, Department of Mines and Natural Resources, Winnipeg, Manitoba, Canada

EUTROPHICATION IN SOME LAKES AND COASTAL AREAS IN FINLAND, WITH SPECIAL REFERENCE TO POLYHUMIC LAKES 48

Pasi O. Lehmusluoto, Assistant in the Department of Limnology, University of Helsinki, Helsinki, Finland

THE RECOVERY PROCESS OF A LAKE WHICH RECEIVED WASTEWATER FROM AN ORE DRESSING PLANT 61

Bengt Ahling, Research Engineer, Swedish Water and Air Pollution Research Laboratory, Stockholm, Sweden

DEPLETION OF OXYGEN BY MICROORGANISMS IN ALASKAN RIVERS AT LOW TEMPERATURES 71

Ronald C. Gordon, Research Microbiologist, Alaska Water Laboratory, Federal Water Pollution Control Administration, College, Alaska, U.S.A.

PREDICTION OF DISSOLVED OXYGEN LEVELS IN THE SOUTH SASKATCHEWAN RIVER IN WINTER 96

Robert C. Landine, Assistant Professor, Department of Civil Engineering, University of New Brunswick, Fredericton, New Brunswick, Canada

Preceding page blank

POLLUTION - A BIOLOGICAL STUDY OF SOME RECEIVING WATERS IN HOKKAIDO 113

Matsunae Tsuda, Professor of Zoology, Zoological Institute, Nara Women's University, Nara, Japan

Toshiharu Watanabe, Zoological Institute, Nara Women's University, Nara, Japan

Kozo Tani, Zoological Institute, Nara Women's University, Nara, Japan

CHEMICAL EFFECTS OF SALMON DECOMPOSITION ON AQUATIC ECOSYSTEMS 125

David C. Brickell, Graduate Research Assistant, Institute of Marine Sciences, University of Alaska, College, Alaska, U.S.A.

John J. Goering, Professor of Marine Sciences, Institute of Marine Sciences, University of Alaska, College, Alaska, U.S.A.

PHOSPHORUS BINDING MECHANISMS DURING SELF-PURIFICATION OF POLLUTED LAKES 139

Jan Werner, Research Chemist, Swedish Water and Air Pollution Research Laboratories, Stockholm, Sweden

CRITICAL REVIEW OF PAPERS ON RECEIVING WATERS 153

Peter A. Krenkel, Chairman and Professor, Department of Environmental and Water Resources Engineering, Vanderbilt University, Nashville, Tennessee, U.S.A.

TREATMENT PROCESSES

THE INFLUENCE OF TEMPERATURE ON THE REACTIONS OF THE ACTIVATED SLUDGE PROCESS 164

Pál Benedek, Head of Water Quality and Technology Department, Research Institute for Water Resources Development, Budapest, Hungary

Péter Farkas, Senior Research Biologist, Department of Water Quality and Technology, Research Institute for Water Resources Development, Budapest, Hungary

TEMPERATURE EFFECTS ON BIOLOGICAL WASTE TREATMENT PROCESSES 180

W. Wesley Eckenfelder, Jr., Professor of Environmental and Water Resources Engineering, Vanderbilt University, Nashville, Tennessee, U.S.A.

Andrew J. Englande, Jr., Research Assistant, Department of Environmental and Water Resources Engineering, Vanderbilt University, Nashville, Tennessee, U.S.A.

EVALUATION OF AERATED LAGOONS AS A SEWAGE TREATMENT FACILITY IN THE CANADIAN PRAIRIE PROVINCES 191

Archie R. Pick, Research Engineer, Water Works and Waste Disposal Division, The Metropolitan Corporation of Greater Winnipeg, Winnipeg, Manitoba, Canada

George E. Burns, Engineer of Design, Water Works and Waste Disposal Division, The Metropolitan Corporation of Greater Winnipeg, Winnipeg, Manitoba, Canada

Dick W. Van Es, Engineer of Sewage Disposal, Water Works and Waste Disposal Division, The Metropolitan Corporation of Greater Winnipeg, Winnipeg, Manitoba, Canada

Richard M. Girling, Assistant Engineer of Design, Water Works and Waste Disposal Division, The Metropolitan Corporation of Greater Winnipeg, Winnipeg, Manitoba, Canada

DESIGN CONSIDERATIONS FOR EXTENDED AERATION IN ALASKA 213

Sidney E. Clark, Acting Chief, Cold Climate Research Program, Alaska Water Laboratory, Federal Water Quality Administration, College, Alaska, U.S.A.

Harold J. Coutts, Research Engineer, Cold Climate Research Program, Alaska Water Laboratory, Federal Water Quality Administration, College, Alaska, U.S.A.

Conrad Christianson, Research Engineer, Cold Climate Research Program, Alaska Water Laboratory, Federal Water Quality Administration, College, Alaska, U.S.A.

CHEMICAL TREATMENT OF MECHANICALLY AND BIOLOGICALLY TREATED WASTEWATER 237

Arne Rosendahl, Sanitary Engineer, Norwegian Institute for Water Research, Oslo, Norway

BIOLOGICAL AND CHEMICAL WASTE TREATMENT EXPERIMENTS IN FAR NORTHERN SWEDEN 252

Peter Balmér, Consulting Engineer, Allmänna Ingenjörsbyrån AB, Stockholm, Sweden

BIOLOGICAL SEWAGE TREATMENT IN A COLD CLIMATE AREA 263

Keichi Koyama, Associate Professor, Department of Sanitary Engineering, Hokkaido University, Sapporo, Japan

Shigeo Terashima, Professor, Department of Sanitary Engineering, Hokkaido University, Sapporo, Japan

Yasumoto Magara, Instructor, Department of Sanitary Engineering, Hokkaido University, Sapporo, Japan

MICROBIOLOGIC INDICATORS OF THE EFFICIENCY OF AN AERATED, CONTINUOUS-DISCHARGE, SEWAGE LAGOON IN NORTHERN CLIMATES 286

John W. Vennes, Professor of Microbiology, School of Medicine, University of North Dakota, Grand Forks, North Dakota, U.S.A.

Otmar O. Olson, School of Medicine, University of North Dakota, Grand Forks, North Dakota, U.S.A.

DISINFECTION AND TEMPERATURE INFLUENCES 312

Cecil Chambers, Research Microbiologist, Biological Research Program, Advanced Waste Treatment Research, Federal Water Pollution Control Administration, Cincinnati, Ohio, U.S.A.

Gerald Berg, Chief, Virology Section, Waste Identification and Analyses Activities, Advanced Waste Treatment Research, Federal Water Pollution Control Administration, Cincinnati, Ohio, U.S.A.

CRITICAL REVIEW OF PAPERS ON TREATMENT PROCESSES 329

Karl Wuhrmann, Professor, Swiss Federal Institute of Technology, Zurich, Switzerland

SYNOPTIC STUDY OF ACCELERATED EUTROPHICATION IN LAKE TAHOE - AN ALPINE LAKE

Charles R. Goldman, Gerald Moshiri
and Evelyne de Amezaga

INTRODUCTION

Lake Tahoe, located at an elevation of 6,229 feet near the crest of the Sierra-Nevada in a graben basin, is among the clearest alpine lakes in the world. Secchi measurements over 35 meters have been recorded and the lake still has an extinction coefficient between 0.047 and 0.061 m^{-1}. Its watershed of 800 Km^2 is small in comparison to its surface of 499 Km^2. The average depth of Tahoe is 313 meters and it contains 156 Km^3 of water with a retention time of about 700 years (Fig. 1).

With the exception of the early observations of Le Conte (1883a, 1883b, 1884) and the limnological reconnaissance of Juday (1907), Kemmerer et al. (1923), and Hutchinson (1937), the lake has received little limnological study until the last ten years. Geological studies of the lake have included fathometry of the basin as well as studies of sediment distribution (Court, Goldman, and Hyne, in press). The general trophic state of Tahoe may be classified as very oligotrophic as compared with a variety of other lake types (Goldman, 1967, 1968). Attached aquatic plants are found to a depth of 100 meters (Frantz and Cordone, 1967) and the crayfish *Pacifasticus leniusculus* is extremely abundant in the littoral zone (Abrahamsson and Goldman, 1970).

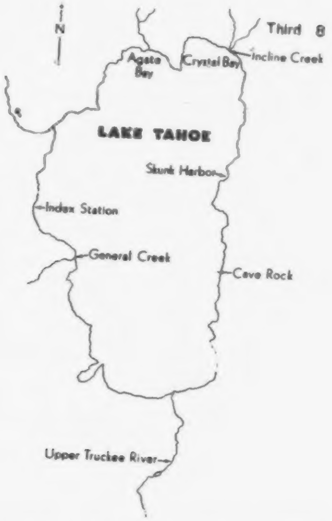


FIGURE 1 Lake Tahoe, California-Nevada

Culture experiments show that the lake is deficient in available iron, that the interaction between nitrogen and phosphorus in Tahoe is very complicated, and that the various tributaries of the lake differ greatly in their stimulation of algal growth (Goldman, 1964; Goldman and Armstrong, 1969; Goldman, Tunzi, and Armstrong, 1969).

The variation in productivity in the lake was first documented during two synoptic studies in 1962 (Goldman and Carter, 1965). A second series of three synoptics was run in 1967, and a third series in 1968 which are reported here.

During the last decade the Tahoe basin, an important recreational area of the Sierra-Nevada, has undergone a very rapid increase in both resident and visitor population, and despite its great volume of low nutrient concentration, the measured increase in primary productivity is already a clear sign of accelerated eutrophication (Goldman and Armstrong, 1969). The senior author believes that if transparency is reduced by inorganic turbidity or increased algal growth a lowering of the lake's heat budget will occur with the possibility of freezing. This hypothesis is currently under investigation since Tahoe does not freeze, and a winter ice cover, by promoting reduced conditions at the mud-water interface, would probably accelerate eutrophication.

The present synoptic study is directed toward identifying the major sources of nutrients reaching Tahoe and the patterns of eutrophication they produce in the lake. Previous studies proved to be particularly useful approaches to the problem of identifying nutrient sources and evaluating their effect on adjacent Tahoe waters. The investigation reported here included primary productivity of phytoplankton and periphyton as well as examination of species composition, biomass and biotic diversity. In addition, benthic invertebrate organisms were collected and their diversity determined.

Synoptic studies are often made at sea where oceanographic vessels collect large numbers of samples in transects or over broad areas. Such collections are limited in their coverage by the time required for sampling each station and the slow speed of the vessels between stations. Further, most of the marine synoptics cover days, weeks, or months, so that time and weather are important variables. Some improvement is anticipated as automated sampling and aerial reconnaissance techniques are improved, but the limitations indicated above remain serious problems. Synoptics covering physical and chemical parameters have been made in the Great Lakes (e.g. Ayers, et al., 1958, Anderson and Rodgers, 1963, Saunders, Trama and Bachmann, 1962). Synoptic surveys of two New Zealand lakes for water chemistry, phytoplankton, and zooplankton distribution were made by Fish and Chapman (1969). For primary productivity measurements, shipboard incubation has long been used for synoptic surveys at sea, and Sorokin (1959) used the technique in Rybinskii reservoir over a period of seven days in June. He found photosynthesis varied in different parts of the lake as much as tenfold during this period.

Because of the photosynthetic variation from day to day, reflecting changes in the composition of phytoplankton, nutrients, or weather, a synoptic covering the shortest possible time period will have the greatest precision for detecting variation in fertility within the system. The synoptic approach of 1968 reported here was probably unique in covering the entire surface layers of a large lake in a single day without disrupting the natural light cycle and with an *in situ* incubation of the primary productivity samples. This eliminated the uncertainties associated with studies that covered longer time periods with various light conditions, or the necessity of using the unnatural light of deckboard incubation. It provided a nearly instantaneous measure of productivity, and phytoplankton composition and biomass over the whole lake.

METHODS

The synoptic approach evolved from the first series of synoptics done in 1962 to the 1968 series reported here. In 1962 eight stations were incubated *in situ* with varying incubation time from station to station. These were corrected to a daily value on the basis of a diurnal series of samples run at an index station. In 1967 five different stations on three different dates were sampled for comparison of primary productivity. Incubations were done in a rotating incubator under constant artificial light at surface water temperature. Samples for each station were taken at two depths where the photometer measurements indicated 75% and 50% transmission of the available surface light.

The following paragraphs describe the methods used in 1968.

Phytoplankton

In the summer of 1968 three synoptic studies were conducted: synoptic one during 16-17 July synoptic two during 11-12 August and synoptic three during 1-2 September.

Thirty stations were selected, twenty-five of which were located along the shore around the lake. Five stations were in the middle of the lake. In areas where it was desired to test for steep gradients in production, three stations were located relatively close together, such as for General Creek, Upper Truckee, Cave Rock, and Incline Creek where one station was chosen at the mouth of the creek and one on each side.

Two boats were used to collect all the water samples within about a six hour time period. All the water samples were collected at night, between approximately 8:30 pm and 2:30 am, and stored in insulated boxes, so that no photosynthesis or sample warming would take place and so as not to expose the organisms to high surface light or disrupt their diurnal rhythm. Four depths were sampled: 0 meters, 5 meters, 10 meters, and 15 meters. Water samples were collected in 125 ml Pyrex bottles for measurements of primary productivity using the carbon-14 method as modified by Goldman (1963). Phytoplankton and water chemistry samples were collected at the same time. All samples were returned to the index station located near Homewood on the west shore. This is near our laboratory where regular *in situ* productivity measurements are taken on a year round basis.

All primary productivity samples were rapidly injected with carbon-14 solution just before sunrise, lowered into the lake, and suspended at the same depths from whence they were collected. The bottles were left to incubate all day and were returned to the laboratory for filtration at sundown; handling of the large number of bottles was therefore accomplished before any occurrence of photosynthesis and after photosynthesis had stopped in all bottles. This provided an equal incubation period for all samples so that the primary productivity measurements for all stations were comparable.

Phytoplankton samples were collected for each station by pooling aliquots from each depth into one sample. The samples were fixed with neutral Lugol's solution and returned to the University of California campus at Davis for identification, counting, and measurement of individuals for determination of their biomass. An inverted microscope was used for this work.

The water chemistry samples for alkalinity determination were taken for each station at the 5 meter depth only, since no significant change with depth down to 15 meters was detected in Lake Tahoe. Carbon available for photosynthesis was calculated from these samples by titration with acid (Saunders, Trama, and Bachmann, 1962). We have since gone to infrared CO_2 analysis which is more rapid.

Periphyton

Seventy-nine sampling stations were chosen around the lake along the shore and samples taken between 28 March 1968 and 18 September 1968. Periphyton samples were collected on glass cylinders of 40 mm lengths and 24 mm O.D. Pyrex held by test tube holders at a depth of five meters. The pieces of glass tubing were cleaned with hot acid and stored in 0.1N Hcl prior to use. The length of time required for the development of significant attached growth varied according to the productivity of the habitat. A week to ten days incubation was found to be adequate at the 5 meter depth in Lake Tahoe. When the periphyton collectors were retrieved, they were stored frozen in individual plastic snap cap vials until combusted for total carbon content. Combustion of periphyton was achieved by placing the tubing in a .005-inch-thick platinum sleeve in an induction furnace (LECO Series 521). This rapid method for the determination of the carbon content of periphyton depends on the measurement of the combusted carbon in an infrared gas analyzer (Armstrong, Goldman, and Fujita, in press).

Benthos

Thirty-nine locations were chosen, evenly distributed over the lake on a grid, plus one station in Emerald Bay and one on a "lakemount" (discovered and described by C.R. Goldman and J. Court, 1968). Sampling was conducted between August 1, 1968, and October 26, 1968 and served as the basis for a study of sedimentation in the lake (Court, Goldman, and Hyne, manuscript). Only one sample was taken from each station using a Shipek sampler, so that a high degree of quantitative reliability cannot be claimed for the collections. The sediment samples collected varied in amount from less than one liter to more than four liters, depending on the substrate, and were washed and sieved before sorting. Lake water was used for the washing, and care was taken not to cause a grinding of the organisms in the washing of the sand and silt. Sieving was graded and fine enough to prevent the passage of even such small organisms as cladocerans and copepods. Following the washing, the samples were preserved in formalin in small vials. The various organisms were then identified and enumerated.

RESULTS AND DISCUSSION

In 1967 the variation in primary productivity was particularly high in August and September. Comparisons were not converted to mgC/m^2/day because they were incubated under constant light not very representative of the lake conditions. Crystal Bay and the South Shore were the highest stations in August and early September respectively (Fig. 2). With the onset of winter storms and mixing, the variation in stations disappeared in late September.

The 1968 synoptics provided a great deal more data than the 1967 studies. For the synoptic at

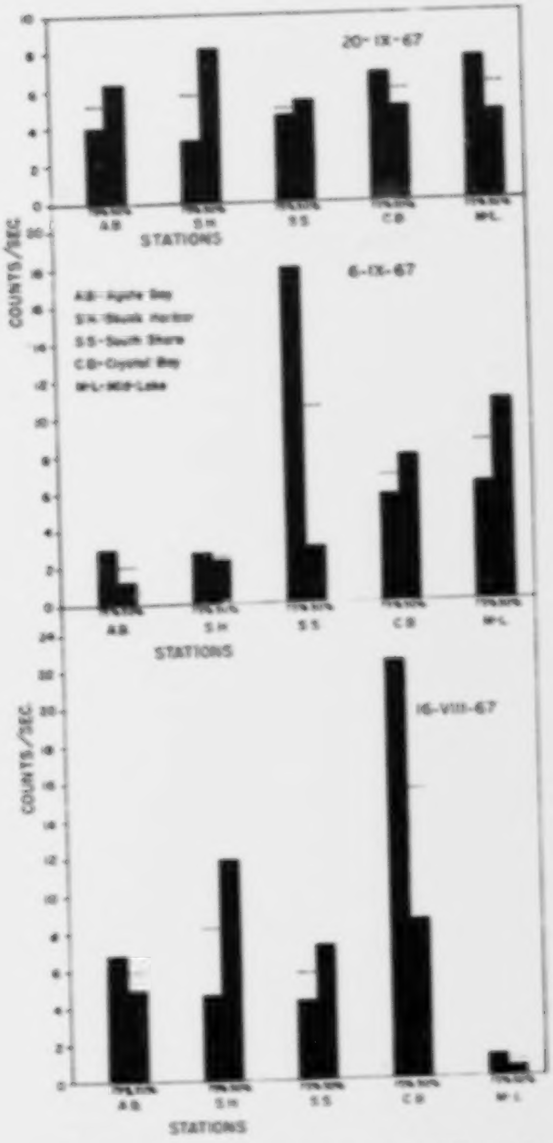


FIGURE 2 Primary productivity measurements taken at five different locations of Lake Tahoe, on three different sampling periods, at the two depths where 75% and 50% light are transmitted.

each station, primary productivity for the day, under a square meter surface, for the 15 meter column of water, was calculated in mgC/m^2/day. A contour map of primary productivity was constructed from this data (Fig. 3). For synoptic one, the range of values was from 11.04 mgC/m^2/day in a midlake station to 78.52 mgC/m^2/day at the mouth of Incline Creek. For the second synoptic, values ranged from 12.41 mgC/m^2/day in a midlake station to 90.86 mgC/m^2/day at Tahoe City. For the third synoptic, values ranged from 17.85 mgC/m^2/day in a midlake station to 84.07 mgC/m^2/day at the mouth of the Upper Truckee River.

In 1968, during each synoptic study, primary productivity was measured at 13 different depths at the index station, and the percentage of the primary productivity of the entire euphotic zone represented by the primary productivity of the first 15 meters was estimated from this comparison. On that basis, the average primary production of the euphotic zone, extended to 105 m (the average depth of the euphotic zone), was found to be 125.0 mgC/m^2/day for the first synoptic, 109.6 mgC/m^2/day for the second, and 193.9 mgC/m^2/day for the third.

For each station, the average of the three synoptics was calculated and a contour map of primary productivity in Lake Tahoe was drawn based on the average of the three synoptics (Fig. 3). The range of average values varied by fourfold, from 14.90 mgC/m^2/day in a midlake station to 57.22 mgC/m^2/day at Tahoe City. The spatial patterns of primary productivity appeared to be not random but ordered. Primary productivity is often high near shore and at the mouths of creeks. It is generally low at stations near the middle of the lake.

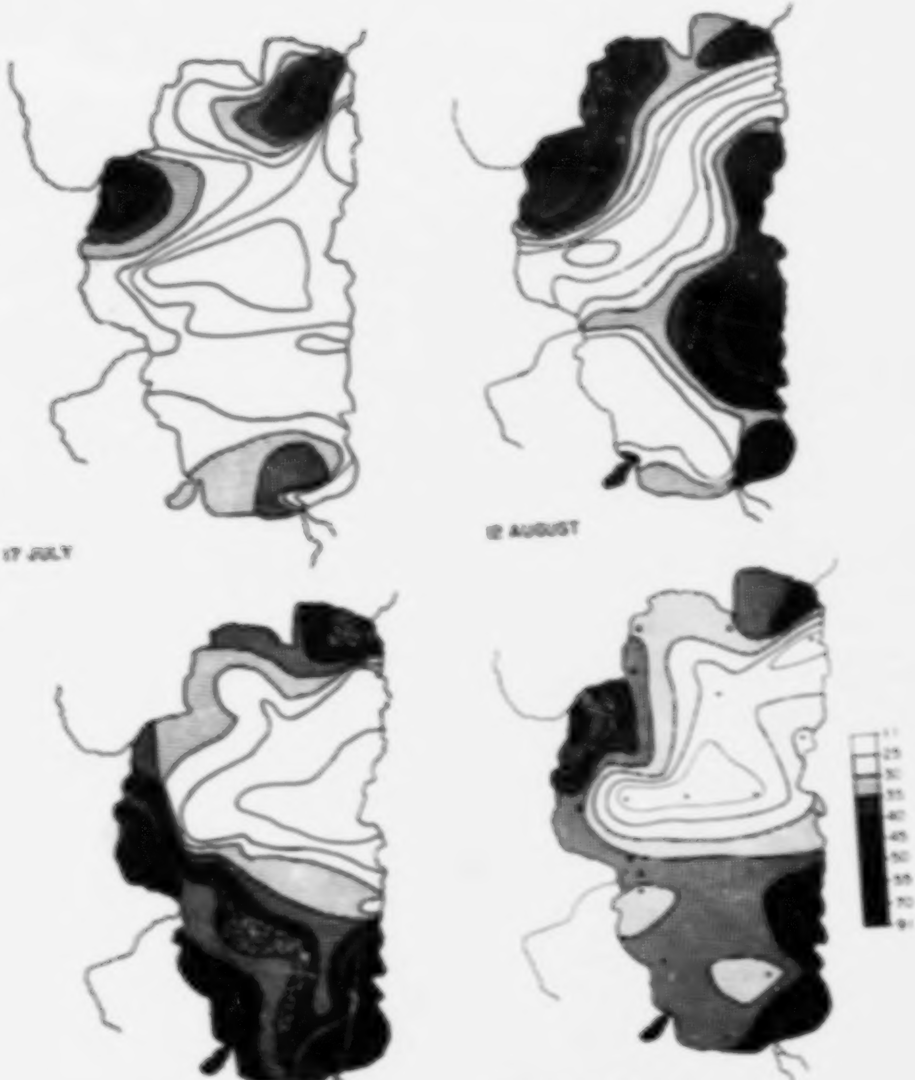
The total number of individual phytoplankton per ml and the total fresh weight (biomass) in milligrams per cubic meter were calculated for each synoptic at each station. Average values for the three synoptics at each station were used to draw isopleth lines of phytoplankton on the map of Tahoe (Fig. 4), as was done for primary productivity. Average values of number of individual per ml varied by 4.5 fold, from 20.58 in midlake to 77.08 at the mouth of Incline Creek and 89.33 at Tahoe City. Average of three synoptics per station for the biomass of the phytoplankton varied by sixfold, from 19.89 mg/m^3 at Rubicon Point and 20.65 in midlake to 77.71 mg/m^3 at the mouth of Incline Creek and 129.56 mg/m^3 at Tahoe City.

Spatial patterns of phytoplankton, like the primary productivity, appeared ordered. Phytoplankton concentration was high near shore and at the mouths of some creeks. It was systemically low in the middle of the lake.

Biotic diversity per individual on the number of individuals per species of phytoplankton was computed for each station and each synoptic. The expression for diversity used here is the information measure of diversity used by Margalef (1957, 1965), Goldman et al. (1968) and Goldman (1970, in press).

$$D = -\sum_{i=1}^{i=m} \frac{n_i}{N} \log_2 \frac{n_i}{N} \qquad 0<D<00$$

where n_i = number of individuals in the i^{th} species
N = Total number of individuals
m = number of species present in the sample



17 JULY

12 AUGUST

2 SEPTEMBER

Average of the 3 Synoptics

FIGURE 3 Contour map of primary productivity in the upper 15 meters of water based on mgC/m^2 /day^1 measured at 30 stations for each of the 3 synoptics. The fourth map is based on values at each station which are the average of the 3 values of the individual synoptics. X's indicate the station locations.

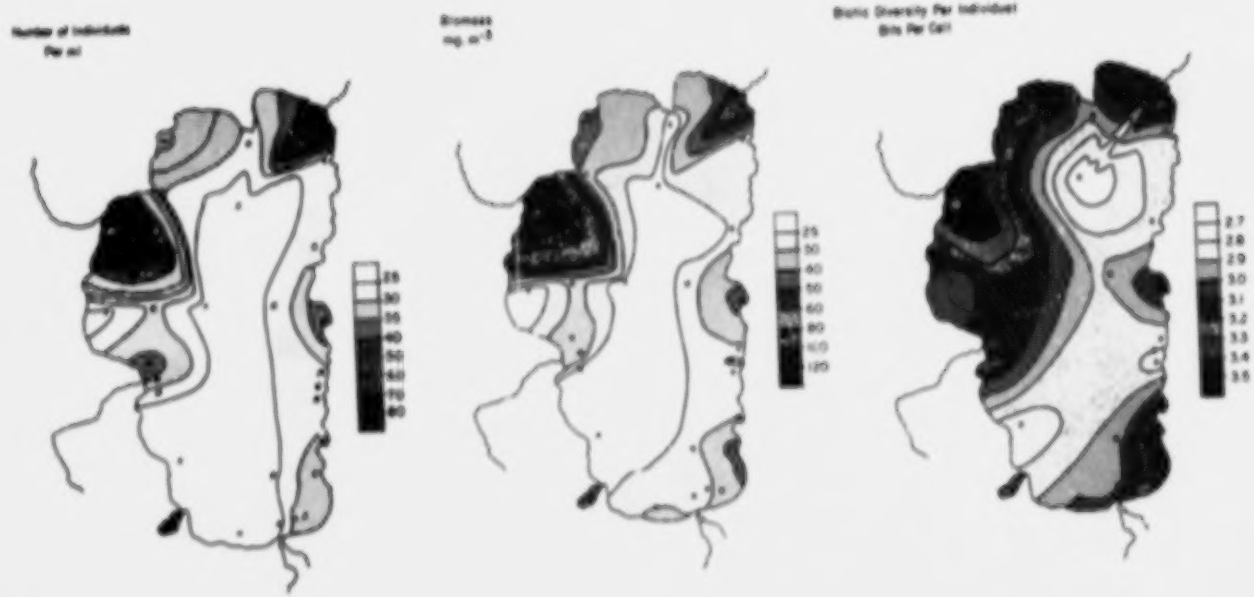

Number of Individuals
Per ml
Biomass
mg m^-3
Biotic Diversity Per Individual
Bits Per Cell
2.7
2.8
2.9
3.0
3.1
3.2
3.3
3.4
3.5

FIGURE 4 Phytoplankton number of individuals, biomass, and diversity per individual. Isopleth lines have been drawn using values at each station which are the average of the 3 individual synoptic values. X's indicate the station locations.

Average values for the three synoptics at each station were used to draw isopleth lines of biotic diversity in bits per individual on the map of Lake Tahoe, as was done for the other variables (Fig. 4). Average values of biotic diversity per individual varied from 2.61 bits per individual in a midlake station to 3.59 bits per individual at Tahoe City. Spatial patterns of diversity appeared to be not random but ordered. Higher values were largely in shallow water areas near shore and at the mouths of some creeks where productivity was also higher. Diversity values were systematically low at some of the deep midlake stations.

A considerable difference between shallow water and deep water productivity and phytoplankton concentration is evident from Figures 3 and 4.

A student *t* test was used to determine the significance of this difference between sampling stations in midlake and those nearer the shore. Four combinations of stations were tested.

I. The four midlake stations of the deep northern part of the lake versus all other stations.

II. The four midlake stations of the deep northern part of the lake plus the midlake deep station offshore from the Upper Truckee River versus all other stations.

III. The four midlake stations of the deep northern part of the lake plus the three stations alongshore off General Creek versus all other stations. General Creek on the West Shore of the lake is a tributary of low nutrient concentration and of small stimulation to algal growth (Goldman and Armstrong, 1969).

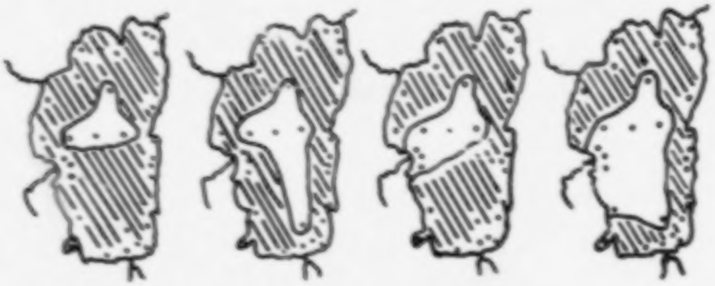
IV. Same as III but including one more station south of General Creek close to shore, and the midlake station off the Upper Truckee River with the other midlake stations, namely 8 stations, versus the other 22 stations.

Differences in primary productivity, concentration of phytoplankton individuals, concentration of phytoplankton biomass, and diversity per individual of phytoplankton were tested. Results are shown in Table 1. Data from all three synoptics and all thirty stations per synoptic (90 data points) were used for each *t* value determination. Significant results were obtained from all four groupings selected for primary productivity, phytoplankton number of individuals, and biomass concentrations. This indicated that the difference between the means of the two groups of stations selected in all four cases was sufficient to warrant the conclusion that the one group with midlake stations was definitely different from the other. Highest significances, 0.1% and 0.5%, were obtained for primary productivity. Grouping arrangements I and II gave the most significant results for primary productivity and grouping IV gave the most significant results for phytoplankton concentration. For the data on phytoplankton diversity, differences between the means of the two particular groups of stations selected in those four cases were not sufficient to conclude that one group was different in diversity from the other. This was probably due to the one midlake station on the northwest of the lake where the diversity was quite high.

These results show the highly significant differences in primary productivity between the shallow and deep water stations of the lake, as well as the difference in phytoplankton numbers and biomass. Furthermore, they show that primary productivity and phytoplankton concentrations in the waters along the shore off the General Creek area were not as different from midlake results as those of the waters sampled offshore near other disturbed, drainage areas. They were more similar

TABLE 1 *t* TEST RESULTS

Significance of Difference Between Sampling Stations in Midlake and Along the Shores



	I Stations 1-25,30 versus 26-29	II Stations 1-25 versus 26-30	III Stations 1,5-25,30 versus 2,3,4,26-29	IV Stations 1,6-26 versus 2-5,26-30
Primary Productivity	4.074	3.906	3.286	3.163
	0.1%	0.1%	0.5%	0.5%
Phytoplankton No. Individuals	2.009	2.317	1.862	2.653
	5%	2.5%	10%	1%
Phytoplankton Biomass	1.706	1.896	1.851	2.470
	10%	10%	10%	2.5%
Phytoplankton Diversity	1.240	1.477	0.388	1.418
	–	–	–	–

Results from all three synoptics on a total of 30 stations were used for each T value determination. Level of significance is for 88 degrees of freedom and is given as percent level under the *t* value.

to the midlake deep water stations than other shoreline waters, since grouping IV also gave significant results, particularly for primary productivity.

Higher diversity (Fig. 4) was evident in the shallower water areas of the lake as well as in the vicinity of streams where productivity was higher due to the inflow of nutrients from the watershed. This pattern of diversity in Lake Tahoe was observed not only in the average values from the three synoptics; each synoptic cruise showed the same general pattern of diversity. The work of Margalef (1964) suggests that the opposite pattern of species diversity should have been found. That is, higher diversity occurring in the less enriched, less productive central waters of the lake and less diversity in the more enriched, more productive shoreward waters of the lake. This position is certainly borne out by our studies of a highly productive lake like Clear Lake, California, which at times may be almost a monoculture of bluegreen algae such as *Aphanizomenon*. However, in highly oligotrophic environments such as Tahoe, a eutrophicating influence such as fertile stream water may increase the diversity in the immediate area. Similarly, benthic forms may enter the lake directly from the streams. The relative importance of these benthic forms to the number of individuals, the biomass of the phytoplankton and directly to the diversity values was determined. Out of the one hundred and three different species of phytoplankton found most commonly in Lake Tahoe, nine were most likely to be benthic forms that became dislodged from their attachment by wave action or otherwise freed from the substrate. These were mainly diatoms and included *Cocconeis placentula*, *Cocconeis discutus*, *Gomphonema parvulum*, *Gomphonema capitatum*, *Cymbella ventricossa*, *Cymbella lanceolata*, *Cymbella cuspidata, and Diatoma vulgare*. Some *Anabaena* have also been found in the benthos of Lake Tahoe. The percentage of those species found in the total phytoplankton population in terms of number of individuals and biomass was calculated at each station for each synoptic study. The same percentages were also calculated from the average values of the three synoptics in number of individuals and biomass at each station. Those results are shown in Figure 5. A comparison of these results with a bathymetric map of the lake does not show a very close correlation between shallow water and high percentage of benthic forms. Conversely, deep waters and low percentage of benthic forms are not closely correlated, although the highest percentages are found in the shallower waters. This may in part be due to the fact that shallow waters are only a small part of a lake whose mean depth is 313 meters. The curious pattern of diversity in Crystal Bay may well reflect the deep canyon running out from shore where benthic forms may not enter the catch so frequently. Wind patterns and water movements might also influence the presence of benthic forms in the phytoplankton population of the different areas of the lake, as might the nutrient history of existing periphyton of the various areas. This is particularly true if we consider that periphyton has only become abundant in the lake during the last ten years as the nutrient content of littoral water has accelerated its growth. Comparing these results with the biotic diversity per individual did not show a systematic relationship. To verify and further investigate this, the biotic diversity per individual was calculated again for each synoptic cruise and for the average of the three, but this time omitting the benthic species. All new values were lower than those obtained from the entire phytoplankton population, but the overall pattern of diversity of the lake remained essentially unchanged.

Rate of growth of periphyton varied from .094 mgC/m^2/day, north of Skunk Harbor between 20 July and 18 August, to 2.228 mgC/m^2/day, 70 yards offshore north of Sand Harbor between 15 July and 23 July. Periphyton growth was found to be a function of both the location and the time of incubation period (Fig. 6).

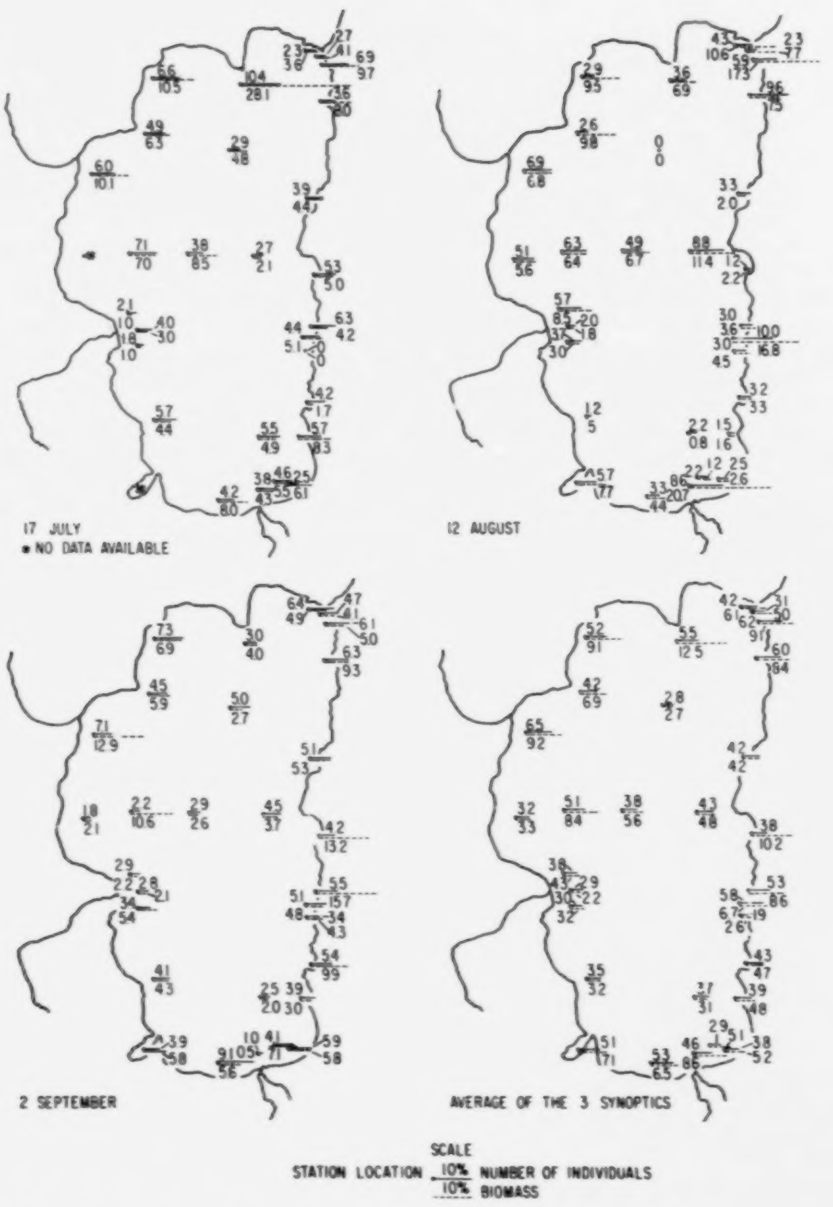

17 JULY
* NO DATA AVAILABLE
12 AUGUST
2 SEPTEMBER
AVERAGE OF THE 3 SYNOPTICS
SCALE
STATION LOCATION
10% NUMBER OF INDIVIDUALS
10% BIOMASS

FIGURE 5 Percent number of individuals and percent biomass of benthic forms of algae found in total phytoplankton counts, for each synoptic study and on the average.

A high initial loss of samples from periphyton stations resulted in considerable variation in effective sampling effort. Later the original surface floats were all replaced with underwater floats tended by scuba divers. As a result of loss of stations, quite a few values for the rate of growth of periphyton were missing from the overall matrix of data which consisted of rows of stations versus columns of incubation periods. In order to be able to determine variations of rate of growth of periphyton between areas along the shores of the lake, three types of computations were made.

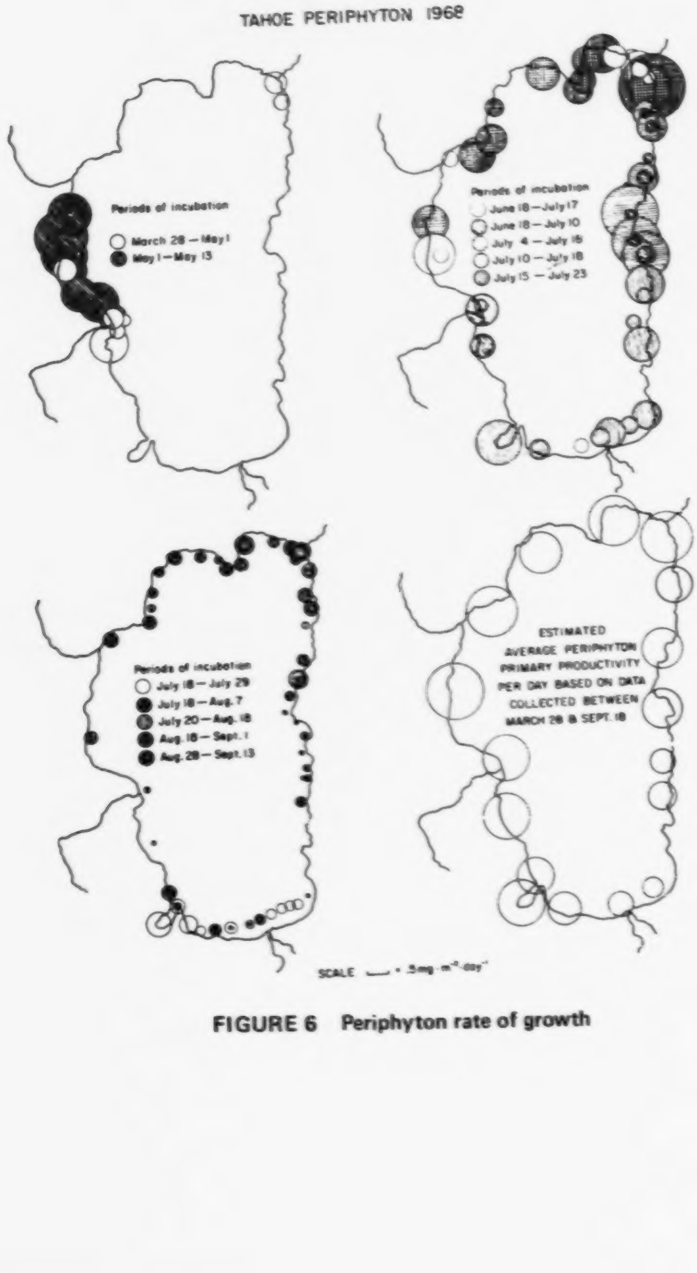

TAHOE PERIPHYTON 1968
Periods of incubation
○ March 28 — May 1
● May 1 — May 13
Periods of incubation
○ June 18 — July 17
○ June 18 — July 10
○ July 4 — July 16
○ July 10 — July 18
● July 15 — July 23
Periods of incubation
○ July 18 — July 29
● July 18 — Aug. 7
● July 20 — Aug. 18
● Aug. 18 — Sept. 1
● Aug. 28 — Sept. 13
ESTIMATED AVERAGE PERIPHYTON PRIMARY PRODUCTIVITY PER DAY BASED ON DATA COLLECTED BETWEEN MARCH 28 & SEPT. 18
SCALE = .5mg·m⁻²·day⁻¹

FIGURE 6 **Periphyton rate of growth**

First, adjacent stations of similar productivity were pooled and averaged to give one value for that particular area for the period of incubation (pooling of rows of the matrix). Seventeen main stations resulted from this sample pooling. Second, periods of incubations adjacent in time and showing similar rate of growth for most stations were pooled to give an average rate of growth for the entire period (pooling of columns of the matrix). Third, missing values for one of the seventeen main stations at one of the resulting five periods of incubation were interpolated from the other values. Interpolations were made favoring rows rather than columns, namely using more points in time than stations, rather than a large number of stations and fewer points in time. After completing the matrix of 17 stations and five incubation periods, columns were averaged to give one average value of rate between 28 March and 18 September 1968 (Fig. 6). These results vary between .343 mgC/m²/day northeast of the Upper Truckee River to 1.010 mgC/m²/day south of Tahoe City.

The average daily amount of growth of periphyton on the bottom of the entire lake was estimated by use of a planimeter for each area between the shore and the 100-meter depth, multiplying the rate of growth of that location by the area found, and summing up the results. The weight of carbon of algae was estimated to be 13% of the total fresh weight of the algae. On that basis the average total fresh weight of periphyton added daily to the bottom of the lake (by photosynthesis on the bottom surface only) was found to be about 576.8 kilograms. This was an underestimate of the bottom average daily growth of periphyton, since no correction was made to account for the irregularity of rock surfaces. Fox, J. L., Odbug, T. O. and Olson, T. A. (1969), found in western Lake Superior that, "After forty-six days of regrowth on artificially denuded rocks in Stony Point Bay, the growth level was approximately eighteen percent of that occurring naturally." Assuming this result valid for Lake Tahoe would have meant that there are about 147,397 kilograms (fresh weight) of periphyton in Lake Tahoe (about 147 metric tons). For purposes of comparison the biomass of planktonic algae was calculated by averaging biomass per square meter for the three synoptic dates, which resulted in an estimate of 2,180 metric tons in the upper euphotic zone of the lake.

Thirty-three different genera representing a total of twelve different classes of animals were identified in the benthos samples (Table 2). Two genera composed 46.6% of all identified organisms. The dominant organism found in 30 of the 41 stations was *Stygobromus* (Amphipoda). A total of 219 of these identified, which represented 27.5% of all the identified organisms and 24.5% of all organisms found. The second dominant organism found in 26 stations was *Candona* (Ostracoda). One hundred and fifty two were found which represented 19.1% of the identified organisms.

The total number of organisms found in any one sample varied from 0 to 79 (Fig. 7). Values of diversity per individual were computed for each station and varied from 0 to 2.94 bits per individual (Fig. 8).

Values of diversity per individual for our data on benthos were probably more relevant than total number of individuals per sample, due to the limited number and difficulty of obtaining samples of comparable size.

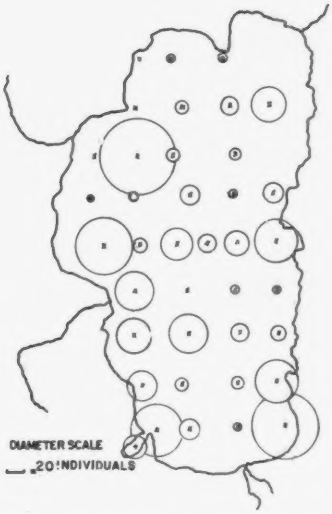

DIAMETER SCALE
20 INDIVIDUALS

FIGURE 7 Lake Tahoe 1968 Benthos - Total Number of Individuals Per Station Per Sample

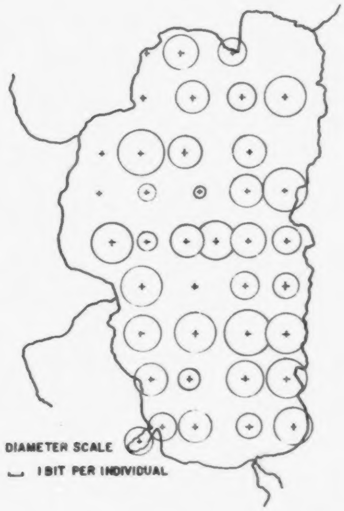

DIAMETER SCALE
1 BIT PER INDIVIDUAL

FIGURE 8 Lake Tahoe 1968 Benthos Diversity Per Individual

TABLE 2

Organisms Found in the Benthos Samples

CLASS	GENERA	NO. OF STATIONS ORGANISMS PRESENT	TOTAL NO. ORGANISMS		% OF TOTAL NO. ORGANISMS
Protoza	*Tetrameris*	3	7		
				7	.8
Turbellaria	*Phagocata*	20	79		
	Dendrocoelopsis	1	3		
	Rhynchoscolex	1	1		
				83	9.3
Nemata	*Trilobus*	8	25		
	Dorylaimus	8	23		
	Iotonchus	8	20		
				68	7.6
Oligochaeta	*Peloscolex*	19	57		
	Lumbriculus	3	26		
	Limnodrilus	4	24		
	Henlea	2	19		
	Ilyiodrilus	5	13		
	Rhynchelmis	3	8		
	Haplotaxis	1	5		
	Naidium (Breviseta)	1	2		
	Eiseniella	1	2		
	Aeolosoma	1	1		
	(Immature)	3	19		
	(Newly hatched)	4	20		
	(Cocoons)	9	24		
	(Eggs)	2	2		
				222	24.9
Ostracoda	*Candona*	26	152		
				152	16.9
Copepoda	*Diaptomis (Tyrelli)*	1	37		
	Epishura	1	3		
				40	4.5
Malacostraca Amphipoda	*Stygobromus*	30	219		
	Hyalella	9	28		
	(Decomposed)	2	2		
	(Unidentified)	2	2		
				251	28.2
Insecta (Chironomidae)	*Orthocladiinae**	6	17		
	Procladius	2	2		
	Chironomus (Atritibia)	1	1		
	Phaenopsectra	1	1		

TABLE 2 (Cont'd)

Organisms Found in the Benthos Samples

CLASS	GENERA	NO. OF STATIONS ORGANISMS PRESENT	TOTAL NO. ORGANISMS		% OF TOTAL NO. ORGANISMS
Insecta (Plecoptera)	*Capina*	3	7		
	(Terrestrial Gnat)	1	1		
	(Midge)	1	1		
	(Larva)	1	1		
	(Diptera Adult)	1	1		
	(Unidentified)	1	1	33	3.7
Acari	*Lebertia*	2	2		
	Limnochares	1	1	3	.3
Mollusca Gastropoda	*Parapholyx*	1	6		
	Gyraulus	1	1	7	.8
Pelecypoda	*Pisidium*	2	4.4		
Terrestrial Arachnid	*Arachnida**	1	1		.1
Miscellaneous	Eggs	9	18		
	Cocoons	1	3	21	2.3
		Total No. of Organisms Found		892	

*Not identified to Genus

CONCLUSIONS

The synoptic approach is very useful in studying cultural eutrophication, as it enables the investigator to accurately locate sources of nutrients before the entire lake has undergone change. Lake Tahoe showed several areas of increased fertility. These were at the South Shore under the influence of the Truckee River drainage and high resident population, in Crystal Bay where Incline Creek and Third Creek drained highly disturbed land, and near the outflow of the lake where there were both a high resident population and fairly extensive areas of shallow water.

Despite the lake's great volume for dilution of the annual inflow, local nutrient sources are altering the productivity pattern around the lake. A spring bloom of algae, which actually turned a path of Tahoe's traditionally blue water green, has already been observed at the south shore and periphyton growth has become luxuriant around the entire lake margin during the last decade. In August 1969, following extensive land disturbance associated with construction of a golf course and a subdivision, Third Creek was found to be extremely stimulating to growth of Tahoe algae. In the experiment summarized in Figure 9, 10% of this stream water added to Tahoe's natural phytoplankton population stimulated photosynthesis by over 600%. When compared with controls, Incline Creek and the Upper Truckee River were also notable lake fertilizers in contrast to relatively undisturbed General Creek. This is not surprising since the Incline Creek drainage has been recently subdivided for homes and the Upper Truckee River and some of its tributaries have also suffered considerable disturbance. Further, this drainage was used for some time as a land disposal site for treated sewage. The sewage is now being exported out of the basin, but the land disposal site still provides a drainage high in nitrogen.

Although occassional high periphyton values are encountered near tributaries, there appears to be less correlation with tributaries than was found for phytoplankton productivity and biomass. In general, distribution of periphyton is fairly uniform around the lake. This probably reflects the steady movement of water over the littoral zone of the lake which distributes the nutrients rather uniformly to these sessile forms.

The results of our study of the possible relationship between diversity and benthic forms can only be considered as a preliminary investigation since our list included only those species that were most likely to be found in the periphyton of Lake Tahoe. Further, more detailed observation of the lake's periphyton species composition at the time of the study would have been a valuable addition.

From the above results we can say that benthic forms of algae contribute overall to increase the diversity per individual of the Lake Tahoe phytoplankton communities, but that they do not appear to be the major determinants of aerial variation in diversity.

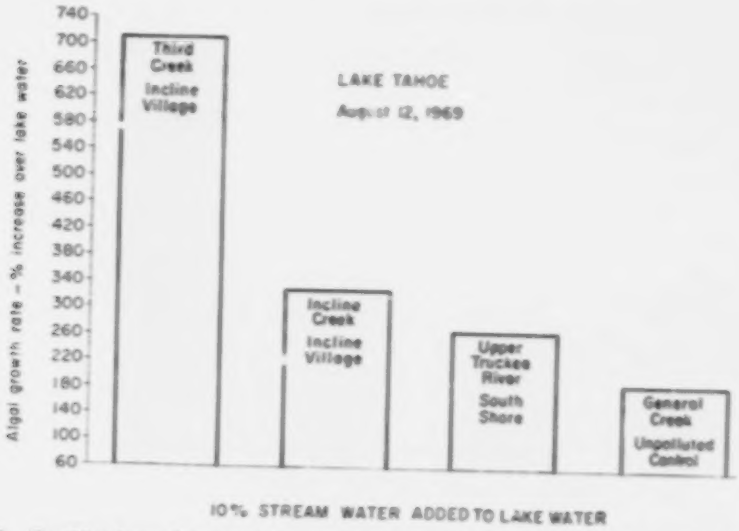


FIGURE 9 **Fertilization of Lake Tahoe water from a 10% addition of four of its tributaries. Algal growth rate was measured with the natural Tahoe population by radioactive carbon uptake. Highest biostimulation was measured in Third Creek following disturbance of its watershed. General Creek, which drains an area of undisturbed land, is used as control.**

The results of the benthic survey are the most difficult to interpret, for there does not appear to be a systematic pattern to either the distribution of numbers of organisms or the distribution of biotic diversity. This condition is especially evident when compared to the synoptic surveys of primary productivity where the primary productivity rate increased sharply near the mouths of some of the creeks. It is quite possible that the lack of pattern in the benthos data may be a result of the lack of replicated samples and the small and variable sample sizes. However, the abundance and diversity of the benthic organisms may not be functions of the same environmental property as the abundance, productivity and diversity of the phytoplankton. Algae are clearly associated with the mouths of creeks and their productivity has been shown to be stimulated by additions of creek water (Goldman and Armstrong, 1969). The environmental factors that affect the benthos have not been determined, and a lack of pattern in benthic numbers and diversity may be an indication that nutrient enrichment is not a primary factor limiting the abundance of various species of benthos.

Tahoe is unquestionably changing. The synoptic approach provides a nearly instantaneous evaluation of conditions as they exist on a given day. Close monitoring of wind and currents could add further to our understanding of water transport in the lake as it affects the presence of more fertile areas. In general, the primary producers must be viewed as a more sensitive indication of increased fertility than chemical parameters, since any additional nutrients appear to move rapidly into the phytoplankton. We characteristically find little or no measurable change in water chemistry (unpublished data) while phytoplankton photosynthesis is showing a very significant change. If the quality of our lakes is to be conserved, synoptic studies can be a great aid in locating and defining the influences of nutrient sources. When combined with bioassays, the relative stimulation of the various sources can be quantified and corrective measures taken before accelerated eutrophication destroys the quality of the environment.

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THE SOUTH BASIN OF LAKE WINNIPEG
AN ASSESSMENT OF POLLUTION

Jo-Anne M. E. Crowe

INTRODUCTION

During their brief geologic life, lakes are subjected to a variety of physical and biological processes which result in their ultimate extinction (Powers and Robertson, 1966; Robertson and Alley, 1966). This is a natural, slow process termed eutrophication. Eutrophication accelerated or augmented by man's activities may be termed pollution.

Classical studies on lake eutrophication include those by Minder (1926), Hasler (1947) and Edmondson, Anderson and Peterson (1956). The most intensive studies on the North American continent have involved the Great Lakes and particularly Lake Erie (Wright, 1933; Brown, 1953; Powers et al., 1959; Beeton, 1960, 1961, 1963 and 1966; Matheson, 1962; Wood, 1963; Carr and Hiltunen, 1965; Matheson, 1965; Powers and Robertson, 1966; Arnold, 1969).

Lake Winnipeg possesses many features which render it comparable with Lake Erie. Both are large Pleistocene lakes of recent origin. In size Lake Erie ranks 13th and Lake Winnipeg 14th among the freshwater lakes of the world. Both are adjacent to major population centers. Both are employed as transportation systems to varying degrees. The commercial fishery of both lakes provides a major source of food and revenue. Finally, both lakes have served as recreational areas.

In contrast to the abundance of published literature dealing with Lake Erie, little material dealing with general limnology is available for Lake Winnipeg. Bajkov (1930) studied the chemical composition of Lake Winnipeg waters and the benthic composition. Studies on the mayflies of Lake Winnipeg were published by Neave (1932). Slack (1967) studied the benthos of certain lakes in Manitoba whose basins were located astride Precambrian and Paleozoic bedrocks. Einarsson and Lowe (1968) studied seiche cycles on Lake Winnipeg and Anon., (1969a) reviewed the problems associated with fluctuating lake levels. An assessment of the possible effects of pollution on the benthos was presented by Cover (1960).

It is worthy to note that Butler (1949) said, "Pollution of waters inhabited by fish is not yet a major problem in Manitoba." It is our contention that this problem has arisen in Lake Winnipeg and that the lake is undergoing a eutrophication cycle similar to that of Lake Erie. The main evidence for increased eutrophication due to pollution is evident in the changes which have occurred in the benthic fauna since 1930.

DESCRIPTION OF THE AREA

Morphology and Morphometry

Lake Winnipeg lies between 50° 22′ and 53° 38′ north Latitude and between 96° 11′ and 99° 00′ west Longitude in the province of Manitoba (Fig. 1). The surface area is approximately 9,230 square miles (23,905.7 km^2).-Thus it ranks as the third largest wholly Canadian lake and the largest in the province of Manitoba.

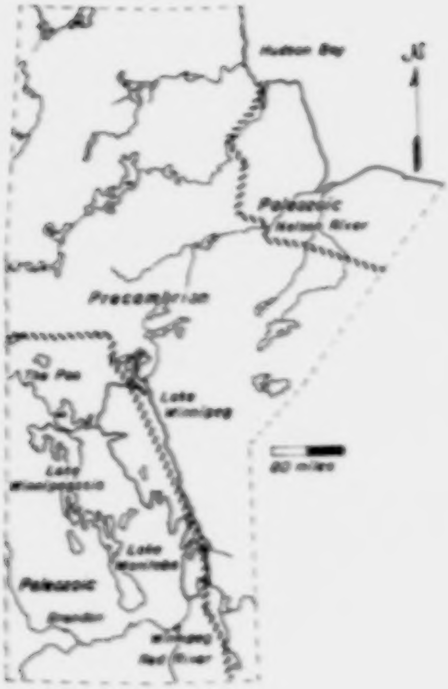
Hudson Bay
N
Paleozoic
Nelson River
Precambrian
The Pas
Lake Winnipeg
60 miles
Lake Winnipegosis
Lake Manitoba
Paleozoic
Brandon
Winnipeg
Red River

FIGURE 1 Province of Manitoba, showing major lakes, rivers, cities and geologic zones

The lake is oriented in a northwest-southeast direction and spans a distance of 253 miles. It is divided naturally into three areas, a north basin with a maximum width of 70 miles and a mean depth of 49 feet (15m), a channel area with a maximum width of twelve miles and a maximum depth of 60 feet (18.2m), and a south basin with a maximum width of 27.5 miles and a maximum depth of 40 feet (12.2m).

Geology

Due to its geologic history and structure, Lake Winnipeg has unique qualities. First, it is a glacial lake of recent origin having assumed its present form approximately 5,000 years ago (Elson, 1958). At present, with neighboring Lake Manitoba and Lake Winnipegosis, Lake Winnipeg remains as the largest remnant of glacial Lake Agassiz (Fig. 1). Second, the contact line between Precambrian and Paleozoic bedrock lies within the basin of Lake Winnipeg adjacent to the east shore (Fig. 1). The latter situation insures that there will be variability both in the composition of the influent waters and the fauna. These aspects have been investigated in this and similar lakes (Rawson, 1953 and 1960; Oliver, 1960; Fedoruk, 1964; Larkin, 1964; Slack, 1965 and 1967).

Watershed and Drainage

The drainage area for Lake Winnipeg is 370,000 square miles (958,300 km^2) extending through portions of four provinces and three states. The major influent rivers are the Saskatchewan from the west and the Winnipeg from the east (Fig. 1). The former contributes 39% of the total drainage receipts and the latter 32% (Anon., 1969a). From the south, the Red River contributes 6% of the total inflow while the remaining 23% is provided by numerous streams and rivers arising in the Precambrian area to the east. Major drainage from the west is scanty and consists primarily of the Saskatchewan River and the Dauphin River (Fig. 1). The lake is drained by the Nelson River to the north into Hudson Bay (Fig. 1). In addition to the major influent sources, considerable water moves between the lake basins as seiches (Einarsson and Lowe, 1968).

Sources of Pollution

There are three potential sources of pollution for the south basin of Lake Winnipeg. The first is run-off from agricultural land to the south and west and is undoubtedly a major contributor (Personal communication, Dr. G. Brunskill, Freshwater Institute, Fisheries Research Board of Canada, Winnipeg).

The second is the city of Winnipeg which is located at the junction of the Red and Assiniboine Rivers approximately 35 miles south of Lake Winnipeg (Fig. 1). Urban population has risen from 29[illegible] in 1931 to 515,661 in 1969 (Personal communication, Mr. A. S. Bready, Metro Inf[illegible]tion, Winnipeg). In the 1930's, all domestic sewage and industrial effluents were discharged untreated in the rivers. This situation has gradually been rectified and at present most municipal and industrial wastes receive treatment in activated sludge plants or aerated lagoons. The total capacity of these systems is 114 million gallons per day. By 1972, all wastes will be treated to effect a 90% removal of B.O.D. and suspended solids (Personal communication, Mr. A. Penman, Waterworks and Waste Disposal Division, Metropolitan Corporation of Greater Winnipeg).

The third potential pollution source is Abitibi Manitoba Paper Limited on the Winnipeg River approximately 6 miles east of Traverse Bay (Fig. 1). The sole product is newsprint produced by a sulfite process. Water consumption in the plant is 11,100,000 Imperial gallons/day. Treatment facilities serve to remove bark and fibre. All effluents from the plant, except domestic sewage, are discharged into the Winnipeg River. B.O.D. values of 7.9 ppm and dissolved oxygen values of 6.7 ppm have been recorded in the river adjacent to the discharge site. Downstream the B.O.D. was 3.1 ppm and dissolved oxygen 10.0 ppm (Personal communication, Mr. D. D. Munro, Abitibi Manitoba Paper Limited, Pine Falls).

METHODS AND MATERIALS

Sampling Procedures

A four-mile grid pattern was adopted for sampling in the south basin of Lake Winnipeg (Fig. 2). Sampling was performed twice each year; in March during the period of winter stagnation and in September, the autumnal overturn. The scope of the sampling schedule from March, 1962 to September, 1969 is summarized in Table 1. Winter sampling was conducted by bombardier and autumn sampling by the Manitoba Fisheries Branch research vessel. The locations of transects were determined by shore landmarks and compass bearings (Fig. 2). Station positions were located by odometer in the bombardier and by a combination of time and engine speed on the research vessel.

Limnological data routinely collected on location included water depth, surface and bottom water temperatures, dissolved oxygen, alkalinity, free carbon dioxide and pH. Turbidity was measured by Seichi disc to the nearest ¼-foot in the ice-free periods. Ice thickness was recorded in the March periods.

TABLE 1

Lake Winnipeg Sampling Schedule, March 1962 to September 1969

Month	Year	Number of Stations	Grid Lines Sampled
March	1962	70	A to N
September	1962	63	A to G
March	1963	22	A, F and K
September	1963	63	A to N
March	1964	40	A to F
September	1964	73	A to N
March	1965	39	A to F
September	1965	66	A to N
March	1966	20	A to D
September	1966	65	A to N
March	1967	16	A to D
September	1967	34	A to N
March	1968	20	A to D
September	1968	33	A to N
March	1969	20	A to D
September	1969	34	A to N

Water depth was measured to the nearest ½-foot by a graduated sounding line. These were confirmed by a model D-11 Bendix depth recorder. Prior to 1967, water temperatures were measured by a brass-encased thermometer in degrees Fahrenheit. Subsequent to this, hydrographic thermometer has been employed. Alkalinity and free carbon dioxide values were determined by standard titration methods (Anon., 1965). pH was determined by comparator using phenol red,

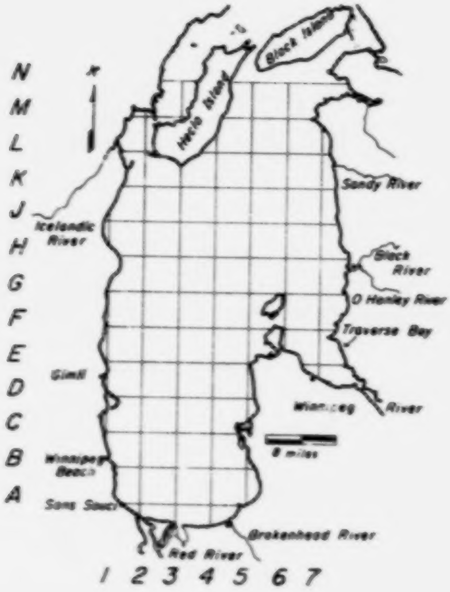
N
M
L
K
J
H
G
F
E
D
C
B
A
N
Hecla Island
Black Island
Sandy River
Icelandic River
Black River
O Hanley River
Traverse Bay
Gimli
Winnipeg
River
8 miles
Winnipeg Beach
Sans Souci
Brokenhead River
Red River
1
2
3
4
5
6
7

FIGURE 2 Four-mile grid sampling pattern, Lake Winnipeg south basin

pH range 6.8 to 8.4, as indicator.

Two-litre bottom water samples were collected by 1,200 cc Kemmerer sampler from predetermined stations during each period. Analyses for a maximum of 20 factors were performed by the Provincial Environmental Health Laboratory in Winnipeg.

During all periods except September 1964 and September 1967, a standard 9-inch Ekman dredge was employed to collect benthos. In September 1964 a Petersen dredge was employed, and in September 1967 a Ponar dredge was employed. The Petersen dredge sampled 0.071 m^2 and the Ponar 0.0518 m^2, as compared with 0.0523 m^2 sampled by the Ekman. Triplicate benthic samples were routinely taken at each station except in March 1962, September 1964, and September 1968. Single dredge hauls were taken in March 1962. Duplicate samples were taken in September 1968 and September 1964, with the exception of four stations in the latter year. At each of these stations six samples were obtained.

Prior to 1967, benthic samples were individually retained in 50-pound capacity polyethylene bags. Whenever feasible, the samples were concentrated by washing through a galvanized bucket equipped with bottom and side screens constructed of U.S. No. 30 mesh (Fremling, 1961). The benthos and remaining debris were placed in four ounce jars containing 10% formalin as preservative. Samples not concentrated in the field were frozen and shipped to Winnipeg where the process was carried out in the Fisheries Laboratory.

Since 1967, an 18-inch diameter hoop net constructed of Nitex nylon has been used to concentrate the samples. The nylon employed has 62 meshes/inch with a mesh opening of 0.0098 inches (0.2450 mm) and is comparable to a No. 50 U.S. screen.

The use of Rose Bengal stain and preservative in the samples after initial concentration has aided in sorting and enumeration of benthos (Mason and Yevich, 1967). This has proven to be more effective than flotation techniques, particularly for nematodes (Ladell, 1936; Lyman, 1943; Birkett, 1957; Anderson, 1959).

Laboratory Procedures

In the laboratory, benthic samples were further concentrated by washing through a No. 50 U.S. bronze screen. The remaining were placed in an enamel pan and scanned by a Luxo light with magnifier to facilitate separation of benthos into taxonomic groups and enumeration. Members of the orders Gastropoda, Ephemeroptera, Amphipoda, Trichoptera, Conchostraca and the families Sphaeriidae and Unionidae were examined with a binocular microscope for taxonomic purposes.

Whole mounts were made of the Nematoda, Chironomidae and Oligochaeta using CMC-10 mounting medium.[1] The heads of the chironomid larvae were severed from the bodies at the thoracic suture and mounted ventral side uppermost. Corresponding bodies were mounted in a lateral position on the same slide (Curry, 1962).

Specialized dissecting techniques were required to display the oligochaete reproductive systems for species identification (Brinkhurst and Cook, 1966). Modifications were made replacing Euparal or Canada balsam with Amman's fluid, a temporary mounting medium (Personal communication, Mrs. M. Moore (Simmons), Zoology Department, University of Toronto).

Where possible, identification was made to genus. Species identification of chironomids and oligochaetes was required. Taxonomic references were as follows: Chironomidae (Johannsen, 1937; Townes, 1945; Johannsen et al., 1952; Robach, 1957; Curry, 1954, 1958 and 1962; Beck and Beck, 1966; Mason, 1968); Ephemeroptera (Neave, 1932; Leonard and Leonard, 1962); Oligochaeta (Brinkhurst, 1963 and 1965; Brinkhurst and Cook, 1966); Gastropoda, Sphaeriidae, Hirudinea, Nematoda, Amphipoda, Conchostraca, Chaoborinae and Trichoptera (Ward and Whipple, 1959).

RESULTS

While other lakes possessing similar morphometric characteristics and geologic structure are termed oligotrophic or at best mesotrophic, Lake Winnipeg exemplifies a eutrophic state (Northcote and Larkin, 1963). Evidence for this is provided by the chemical composition of its water, benthic standing crop, benthic composition and commercial fish production.

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[1] Mention of commercial sources does not constitute endorsement by Manitoba Department of Mines and Natural Resources.

Physical and Chemical Data

Mean values and ranges for chemical data compiled from March 1962 to September 1969 are presented in Table 2. Detailed results for each sampling period are provided in Appendix 1. South-north gradients for all factors except pH have been noted. Values recorded at stations on the A grid line are four times greater than the mean for that period and as high as 30 times greater than the minimum values recorded (Appendix 2; Fig. 2). Similar west east gradients occur for factors such as calcium, magnesium sodium, total hardness, bicarbonate, carbonate, chloride, sulphate and total solids. Individual values along grid 1 are twice as large as those along grid lines adjacent to the east shore (Fig. 2). For the period March 1962 to September 1969, maximum values for most chemical components were recorded during 1966 and 1967 (Appendix 1).

Dissolved oxygen values normally represent saturations of 80% or greater during all periods (Table 2; Appendix 2). B.O.D. values are consistently lower than dissolved oxygen values (Table 2). No decrease in oxygen concentration occurs as water depth increases. No permanent thermocline forms and surface water temperatures are normally less than 2° F higher than those at the bottom.

Bottom deposits over the portion of the south basin under study consist of a sand-silt-clay mixture in the following mean proportions: sand - 3%, silt - 31% and clay - 66% (Personal communication, Mr. W. Michalyna, Canada Department of Agriculture, Winnipeg). Highest proportions of sand were found at stations E-1 and N-5 (Fig. 2). The highest proportion of silt was found at station A-3 and clay at stations F-3, G-3 and H-3 (Fig. 2). A definite south-north gradient could be detected with respect to the silt component. Values for silt on the A grid line are from two to three times the mean level and values of those on grid lines L, M and N (Fig. 2).

Benthos

Values for benthic standing crop are based on 102, 105 and 111 samples taken in the south basin of Lake Winnipeg during June, July and August, 1967, respectively. The three dry weights, including mollusc shell, were 88.18 kg/ha, 56.7 kg/ha and 52.24 kg/ha. Mean standing crop was 65.7 ± 3.93 kg/ha.

Between March 1962 and September 1969, the major benthic groups were Chironomidae 40%, Gastropoda 15%, Oligochaeta 14%, Ephemeroptera 13% and Sphaeriidae 4% (Table 3). The group identified as "others" included representatives from six orders and two families which, combined, comprised a lower percentage than any single group (Table 3). From March 1968, due to increased numbers, it was necessary to consider the Nematoda as a separate group.

Members of four subfamilies were among the Chironomidae identified (Appendix 2). A total of 41 genera and species were identified; 24 (59%) belonged to the subfamily Chironominae, 10 (24%) to the subfamily Tanypodinae, 6 (14%) to the subfamily Orthocladiinae and one (3%) to the subfamily Diamesinae.

Two species of mayfly nymphs, *Hexagenia limbata occulta* (Walker) and *Hexagenia rigida* (McDunnough) were identified (Neave, 1932). 94% of the nymphs were *H. limbata* and 6% were *H. rigida*. The ratio of females to males was 4:1.

The majority of Oligochaeta identified belonged to the family Tubificidae. The species identified,

TABLE 2

Ranges in Values for Chemical Data (mg/l), Lake Winnipeg South Basin, 1930 and March 1962 to September 1969 (Mean Values in Brackets).

	1930 (Bajkov)		March 1962 to September 1969	
True Color Pt Co Units	-		15 to 100	(30)
Apparent color Pt Co Units	-		20 to 70	(45.5)
Turbidity	-		3 to 130	(21.7)
pH	-		6.60 to 8.99	(7.79)
Calcium	13.7 to 62.8	(25.2)	14.4 to 60.0	(28.3)
Magnesium	2.75 to 33.7	(12.1)	0.28 to 21.9	(10.5)
Sodium	-		2.0 to 18.8	(9.4)
Iron	0.14 to 1.6	(0.79)	0.1 to 3.9	(0.79)
Total hardness	-		41 to 224	(111)
Phosphate (ortho)	-		0 to 1.3	(0.21)
Alkalinity	-		1.8 to 226	(96.2)
Bicarbonate	-		40 to 176	(89.1)
Carbonate	28.2 to 128.1	(56.4)	0 to 4.8	(1.2)
Chloride	0 to 27.5	(8.4)	1.0 to 19.8	(7.6)
Carbon dioxide	-		0.1 to 44	(6.4)
Dissolved oxygen	-		0 to 17.7	(11.0)
B. O. D.	-		0.4 to 8.4	(2.86)
Sulphate	3.9 to 118.5	(32.4)	3 to 60	(31.8)
Spec cond μmhos	-		110 to 580	(259)
Total solids	100 to 480	(189.3)	38 to 566	(173)
Suspended solids	-		1.5 to 62	(12.8)
Dissolved solids	-		74 to 374	(177)

in order of occurrence, were *Limnodrilus hoffmeisteri* (Claparède), *Tubifex tubifex* (Müller), *Tubifex kessleri* (Hrabe) and *Limnodrilus udekekiamus* (Claparède). A few specimens, tentatively identifed as *Peloscolex multisetosus* (Smith) were obtained. The other family represented was Lumbriculidae which constituted less than one percent of all oligochaetes. The species identified was *Limbriculus variegatus* (Müller). During some sampling periods, immature forms comprised 50% of all oligochaetes taken. Identification, based on number, size and shape of setae, permitted generic recognition.

TABLE 3

Composition of Benthos, Lake Winnipeg South Basin, March 1962 to September 1969
(numbers in brackets represent percents of total numbers for that period).

Month	Year	Number of Samples	Chironomidae	Ephemeroptera	Oligochaeta	Gastropoda	Sphaeriidae	Mematoda	Others[1]	Total Numbers	Mean Numbers /m^2
March	1962	68	1053 (61)	288 (17)	263 (15)	29 (2)	52 (3)		34 (2)	1719	483
Sept.	1962	147	835 (48)	184 (11)	397 (23)	1 (<1)	208 (12)		109 (6)	1734	225
March	1963	42	576 (78)	44 (6)	2 (<1)	2 (<1)	18 (2)		92 (12)	734	332
Sept.	1963	186	1057 (15)	1915 (28)	1153 (17)	1598 (23)	986 (14)		112 (3)	6821	701
March	1964	96	1085 (39)	417 (15)	1 (<1)	1014 (37)	213 (9)		21 (<1)	2777	552
Sept.[2]	1964	150	353	713 (21)	270 (8)	1473 (44)	523 (16)		26 (<1)	3358	315
March	1965	179	207 (10)	449 (22)	0	1151 (58)	162 (8)		15 (2)	1984	315
Sept.	1965	202	3498 (43)	573 (7)	786 (10)	2182 (27)	930 (12)		109 (1)	8078	765
March	1966	60	396 (57)	79 (11)	65 (9)	143 (20)	0		7 (3)	690	224
Sept.	1966	169	5889 (74)	779 (10)	811 (10)	4 (<1)	329 (4)		107 (2)	7919	801
March	1967	48	620 (66)	28 (3)	176 (19)	12 (1)	89 (10)		15 (1)	940	374
Sept.[3]	1967	99	748 (45)	339 (21)	189 (12)	133 (8)	232 (14)		5 (<1)	1646	318
March	1968	60	1409 (30)	39 (<1)	375 (8)	114 (3)	0	2473 (54)	190 (11)	4600	1464
Sept.	1968	64	622 (13)	154 (3)	1359 (27)	112 (4)	402 (8)	2196 (44)	24 (<1)	4969	1468
March	1969	57	1700 (17)	45 (<1)	265[illegible] (29)	518 (5)	673 (6)	4206 (42)	19 (1)	9924	3225
Sept.	1969	105	1488 (39)	295 (2)	1524 (31)	522 (11)	709 (14)	339 (7)	106 (5)	4983	875
Mean			(40)	(13)	(14)	(15)	(4)	(37)[4]	55	3312	776

1 = *Includes Trichoptera, Hirudinea, Amphipoda, Unionidae, Chaoborinae, Conchostraca and Nematoda, 1962 to 1967.*

2 = *Petersen dredge employed*

3 = *Ponar dredge employed*

4 = *based on percents from 4 years*

The Gastropoda were composed of five families, Physidae, Lymnaeidae, Planorbidae, Valvatidae and Bulimidae. The genera identified were *Physa* (Draparnaud), *Aplexa* (Fleming), *Lymnaea* (Lamarck), *Gyraulus* (Charpentier), *Helisoma* (Swainson), *Valvata* Müller) and *Amnicola* (Gould and Haldeman). The genera *Amnicola*, *Physa* and *Valvata* together composed 90% of the gastropods.

Three genera, *Sphaerium* (Scopoli), *Musculium* (Link) and *Pisidium* (Pfeiffer) had equal representation among the Sphaeriidae. All Trichoptera were tentatively identified as the genus *Limnephilus* (Leach) and all Nematoda as the genus *Dorylaimus* (Dujardin). Members of the family Unionidae identified were *Anodonta grandis* (Say), *Anodonta sp.* (Lamarck), *Lampsilis siliquoidea* (Barnes), *Actinonais carinata* (Barnes) and *Anodontoides ferusaciana* (Lea). The Hirudinea included the species *Placobdella rugosa* (Verrill), *Helobdella stagnalis* (Linnaeus), *Macrobdella decora* (Say) and *Erpobdella* (Blainville).

All Conchostraca were identified as *Caenestheriella* prob. *mexicana* (Claus) and the Amphipoda as *Gammarus*. Occasionally, amphipods resembling *Pontoporeia* were found (September, 1964). Their exact identity is in doubt. It is feasible that these were errant migrants from the north basin displaced by seiche action.

Fish Production

On April 20, 1970, a directive was issued by the Minister of Mines and Natural Resources ordering closure of the summer commercial operations on Lake Winnipeg due to mercury contamination. Mercury levels in excess of the accepted value of 0.5 ppm were found in walleye, northern pike, sauger and perch (Personal communication, Dr. G. Bligh, Freshwater Institute, Winnipeg). Chloralkali plants outside of Manitoba are believed to be the cause for this contamination.

The commercial fishery of Lake Winnipeg is the largest in the province of Manitoba in terms of production, species, value and manpower (Anon., 1969b). Mean annual production from 1931 to 1969 was 13,672,371 pounds (Fig. 3). The major species by catch and value are sauger, *Stizostedian canadense* (Smith), walleye, *Stizostedion vitreum*, whitefish, *Coregonus clupeaformis* (Mitchill) and tulibee or cisco, *Coregonus artedii* complex. Average annual production for these species was 4,494,735, 3,367,026, 2,311,534 and 1,549,578 pounds respectively. Other species taken are white perch, *Roccus americanus* (Gmelin), bullheads, *Ictalurus nebulosus* (Lesueur), carp, *Cyprinus carpio* (Linnaeus), catfish, *Ictalurus punctatus* (Rafinesque), burbot or ling, *Lota lota* (Linnaeus), pike, *Esox lucius* (Linnaeus), goldeye, *Hiodon alosoides* (Rafinesque), perch, *Perca fluviatilis* (Linnaeus), sucker, *Catostomus commersoni* (Lacepede), sturgeon, *Acipenser fulvescens* (Rafinesque) and trout, primarily rainbow, *Salmo gairdneri* (Richardson). Since 1961, there has been a marked decrease in fishing success (Fig. 3).

Hydrographic Data

Mean monthly levels for Lake Winnipeg, 1930 to 1969, are shown in Figure 4. Since 1961, just prior to the initiation of the present monitoring program, lake levels were below the 56 year average. Since 1961, there was an upward trend in water levels which culminated in the record level in 1966. At the end of July 1966 the lake was 717.6 feet above M.S.L. This was more than three feet higher than the normal for that time of year (Anon., 1969a). The rise continued and on September 1, 1966 strong north winds raised the level to 721 feet in the south basin (Anon., 1969a).

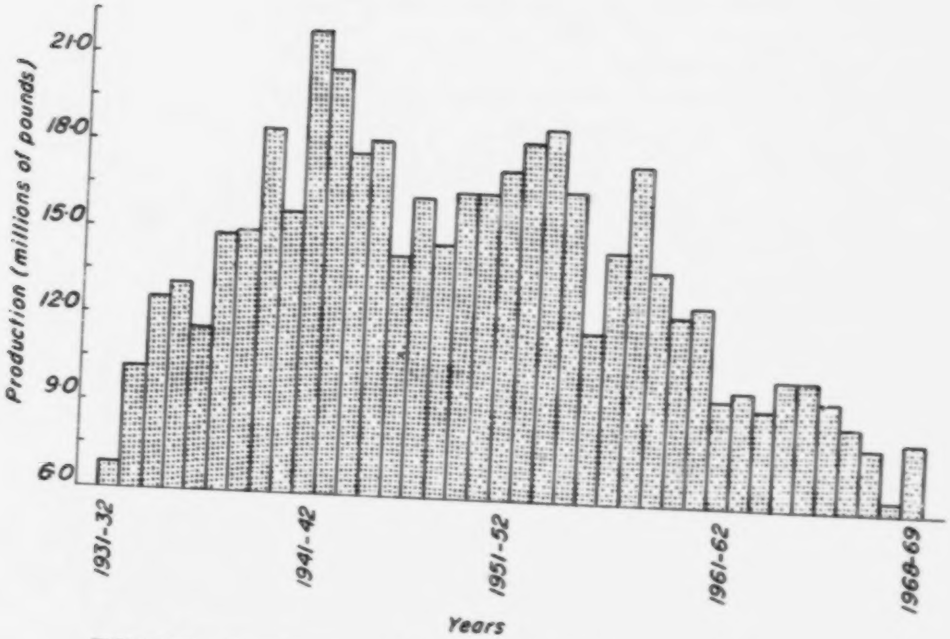

FIGURE 3 Annual commercial fish production, Lake Winnipeg 1962-1969

OBSERVATIONS

On the North American continent, Lake Erie is an authenticated example of a lake whose environment is deteriorating as a result of pollution (Beeton, 1963; Arnold, 1969). Three indices of pollution - chemical and physical data, benthic composition and densities, and commercial fish production - will be examined in Lake Winnipeg and the results compared with those for Lake Erie.

Physical and Chemical Data

Major increases in calcium, magnesium, combined potassium and sodium, sulphate, chlorides and total solids in Lake Erie between 1906 and 1958 are well documented (Lewis, 1906; Dole, 1909; Leverin, 1947; Thomas, 1954; Fish, 1960; Beeton, 1961 and 1963). Turbidity measurements have been conducted and increases noted (Chandler, 1940, 1942a and b, 1944; Powers et al., 1959; Fish, 1960; Kramer, 1961). Evidence supplied by dissolved oxygen determinations supports the hypothesis of severe oxygen depletion over large portions of the Lake Erie basin (Britt, 1955a; Wright, 1933; Powers et al., 1959; Fish, 1960; Beeton, 1961 and 1966; Carr, 1962; Thomas, 1963). Beeton (1961) states that there has been a general warning trend in the waters of Lake Erie.

Phosphates and nitrates have been charged as the major factors producing accelerated eutrophication in Lake Erie. Direct measurements, input rates and increased levels have been recorded (Lewis, 1906; Chandler and Weeks, 1945; Wright, 1933; Curl, 1957; Beeton, 1961; Kramer, 1961; Matheson, 1962; Verduin, 1964; Harlow, 1966; Ownbey and Kee, 1967).

The only ionic component in Lake Winnipeg showing an increase between 1930 and the period 1962 to 1969 was calcium (Table 2). Values for calcium, iron, chloride and total solids were higher in 1969 than in 1930 and maximum values for most components were recorded in 1966 and 1967 (Appendix 1). Seichi disc transparencies have declined from one to two meters in 1930 to less than one meter in 1969 (Bajkov, 1930). Due to the shallow depth of the basin and constant mixing by wind action, no oxygen depletion results. The variation in chemical concentrations would appear to be more closely correlated with climatic factors (i.e., precipitation, discharge rates and lake levels) than loading rates for pollutants (Fig. 4). The absence of any apparent increases in the concentrations of chemical components in Lake Winnipeg is probably due to the length of residence time and discharge rates. Residence time for Lake Winnipeg has been estimated at from four to ten years (Personal communication, Dr. G. Brunskill, Freshwater Institute, Fisheries Research Board of Canada, Winnipeg). Approximately 30,600 tons of total dissolved solids are discharged from Lake Winnipeg each day. This value is based on a mean T.D.S. value of 177 ppm and a mean flow of 59,200 cfs on the Nelson River.

Benthos

Evidence for the eutrophic nature of Lake Winnipeg is provided by both chemical data and benthos (Tables 2 and 4). Mean standing crop of benthos was calculated as 65.7 ± 3.93 kg/ha. When this is corrected to deduct the weight of mollusc shells, the value is 58.43 kg/ha. Lakes lying on the margin of the Precambrian Shield have standing crops of from 7.1 to 9.1 kg/ha (Rawson, 1960). Northern oligotrophic lakes have standing crops ranging from 1.6 to 4.7 kg/ha while lakes south of the Precambrian Shield normally have standing crops greater than 9.1 kg/ha (Table 4). The standing crop of benthos in Lake Winnipeg more closely approximates values determined by Lundbeck (1926) and guarantees its classification as eutrophic. The disparity between values based on Bajkov's data (1930) and present calculations could be indicative of eutrophication. Areas sampled and the number of samples probably explain the lower values obtained by Slack (1967).

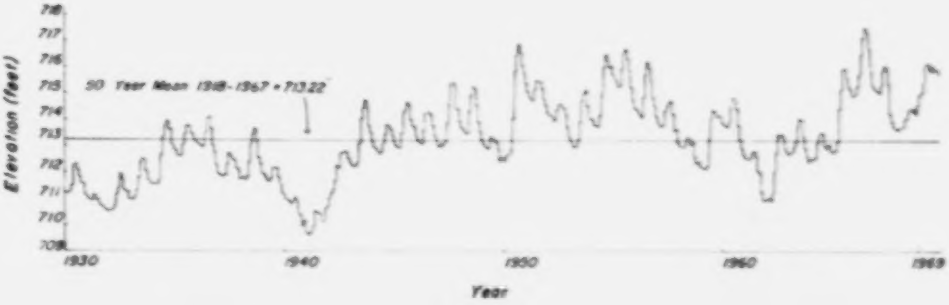

FIGURE 4 Hydrographic data, Lake Winnipeg 1930-1969

The densities of benthos in 1930 and the period 1962 to 1969 provide additional proof of eutrophication. Figures based on Bajkov's (1930) data indicate that benthic densities varied from 437 to 578 organisms/m^2. Present data indicates densities of 776 organisms/m^2 (Table 3).

In 1930, the amphipods formed from 44% to 85% of total benthic numbers based on certain stations which correspond to the present sampling program (Bajkov, 1930). This feature Lake Winnipeg shared with Lake Athabasca and Great Slave Lake (Larkin, 1948; Rawson, 1953). At these same stations the mayflies formed 40% and the midge larvae 6% of total benthic numbers. Oligochaetes did not contribute significantly to total benthic numbers. The present data indicates a decline in amphipod numbers. Decline in both the amphipods and chironomids seems to be correlated with an increase in both the chironomids and oligochaetes.

The effects of pollutional substances on benthos is manifested by a decline in the numbers of species, a decline in the densities and numbers of those species intolerant of pollution, and a corresponding increase in the numbers of those tolerant species (Keup, Ingram and Mackenthun, 1966). Both the amphipods and mayflies are forms known to be intolerant of pollution; their use as biological indicators of water quality is recognized (Wright and Tidd, 1933; Campbell, 1939;

TABLE 4

Characteristics of Certain Canadian Lakes.

Lake	Area km^2	Mean Depth (m)	Total Dissolved T D S ppm Solids	Dry Weight of Benthos kg/ha
Cree[1]	1,152	14.9	32	1.6
Wollaston[1]	2,062	20.6	31	4.7
Reindeer[1]	5,569	17.9	61	1.6
Athabasca[2]	7,770	26.	58	4.1
Great Bear[3]	28,490	>100.	99	0.5
Great Slave[8]	27,200	62	150	2.5
Lac la Ronge[1,4]	1,178	12.7	179	2.9[5]
Amisk[1]	321	13.2	130	9.1
Athapapuskow[7,4]	252	30.[9]	-	3.4 ± 1.4
Reed[7,4]	190	-	109	2.4 ± 1.0
Winnipeg[6]	23,906	13.[9]	177	44.[1]
Winnipeg[7,4]				31.2 ± 12.5
Falcon[7,4]	15	7.6[9]	78	3.6 ± 1.5
Waskesiu[1]	70	11.1	188	24.6
Ile a la Crosse[1]	446	9.2	172	9.0

1 = *Rawson, 1960*　*2* = *Larkin*　*4* = *Rawson, 1951*

5 = *Oliver, 1960*　*6* = *Bajkov, 1930*　*8* = *Rawson, 1953*

9 = *approximate*　*10* = *Estimated*

Gaufin and Tarzwell, 1952 and 1956; Britt, 1955a and b; Gaufin, 1957 and 1959; Beeton, 1961; Leonard, 1962; Wood, 1963; Beak, 1964; Fremling, 1964; Carr and Hiltunen, 1965; Henson, 1966; Hiltunen, 1969; Grimås, 1969; Arnold, 1969). In the south basin of Lake Winnipeg, the amphipods have declined from a mean value of 65% to less than 1% of total benthic numbers and the mayflies from 30% to 13% between 1930 and the period 1962 to 1969 (Bajkov, 1930; Table 3). Even more striking is the absence of mayfly nymphs from grid lines A, B and C (Fig. 2). This would seem to mark the point of most northerly penetration of pollutants from the Red River.

The Oligochaeta, particularly members of the family Tubificidae, are known to thrive under conditions of organic enrichment commonly encountered in polluted environments. Oligochaete densities and species representation have been used as indices of pollution (Wright, 1933; Britt, 1955a and b; Beeton, 1961; Carr and Hiltunen, 1963; Henson, 1966; Johnson and Matheson, 1968; Hiltunen, 1969). Oligochaete densities in the south basin of Lake Winnipeg are normally larger than the 1400/m^2 along the A grid line and decrease to 275/m^2 along the B grid line (Fig. 2). Using Wright's system (1933), the density of 1400/m^2 would be indicative of moderate pollution.

Approximately 95% to 98% of all oligochaetes in the south basin of Lake Winnipeg were the

TABLE 4 (Cont'd)

Characteristics of Certain Canadian Lakes.

Lake	Major Benthic Groups	Geology of Basin	Flushing Time (Years)
Gree[1]	Chironomidae	Precambrian	-
Wollaston[1]	Amphipoda	Precambrian	14
Reindeer[1]	Amphipoda	Precambrian	11
Athabasca[2]	Amphipoda and Chironomidae	Precambrian	-
Great Bear[3]	Mollusca	Precambrian and Paleozoic	-
Great Slave[3]	Amphipoda	Precambrian	-
Lac la Ronge[1,4]	Chironomidae	Precambrian	19
Amisk[1]	Amphipoda	Precambrian	3
Athapapuskow[7,8]	Amphipoda	Precambrian	-
Reed[7,8]	Chironomidae	Precambrian	-
Winnipeg[9]	Amphipoda	Precambrian	4 to 11 [10]
Winnipeg[7,8]	Amphipoda and Ephemeroptera	Precambrian	-
Falcon[7,8]	Chironomidae	Precambrian	-
Waskesiu[1]	Chironomidae	Paleozoic	-
Ile a la Crosse[1]	Chironomidae, Amphipoda and Sphaeriidae	Paleozoic	0.75

species *Limnodrilus hoffmeisteri* (Claparède), *Limnodrilus sp.* and *Tubifex tubifex* (Muller). These commonly occur in polluted zones (Johnson and Matheson, 1968; Hiltunen, 1969).

The last group of benthic organisms commonly occurring in Lake Winnipeg and known to be pollution-tolerant is the Chironomidae (Britt, 1955a and b; Beeton, 1960 and 1961; Wood, 1963; Carr and Hiltunen, 1965; Keup, Ingram and Mackenthun, 1965; Hiltunen, 1969). Brenniman, Soyak and Curry (in press) have classified chironomic species according to water quality preference. This method was adopted and modified for use in Lake Winnipeg. All species of the genera *Chironomus*, *Chironomus (Kiefferulus)*, *Chironomus (Einfeldia)* and *Glyptotendis* were classified as "pollutional". The genera *Pentaneura*, *Polypedilum*, *Harnischia*, *Calopsectra* and *Cryptochironomus* were classified as cosmopolitan and the remainder of the species as "others", i.e., their water quality preference is unknown. No species classified as "clean water" by Curry were identified in Lake Winnipeg. There is a distinct possibility that species classified as "others" may, in reality, be clean water forms. However, our data indicates that 39% of the chironomids could be classified as "pollutional".

The trophic level for Lake Winnipeg was tentatively determined as 1.5 (Brinkhurst, Hamilton and Herrington, 1968). This is less than that determined for western lake Erie (2.0).

TABLE 5

Classification of Chironomidae According to Water Quality Preference Lake Winnipeg South Basin, September 1962 to September 1969 (values are percents of total chironomid numbers for that period)

Year	Month	Pollutional	Cosmopolitan	Others
1962	September	54	25	20
1963	March	34	46	20
	September	26	53	21
1964	March	17	71	12
	September	79	14	7
1965	March	4	66	30
	September	43	44	13
1966	March	26	44	30
	September	66	20	14
1967	March	43	45	12
	September	55	31	14
1968	March	27	67	6
	September	22	42	36
1969	March	44	48	8
	September	40	50	10
Mean		39	44	17

In conclusion, it is evident by changes in the benthic composition, that Lake Winnipeg has been subjected to eutrophication believed to be the result of pollution. These changes are similar to those which have occurred in Lake Erie and which are presently occurring in Lake Michigan (Powers and Robertson, 1965; and Robertson and Alley, 1966). The first change in benthic composition was a shift from Amphipoda to Chironomidae. Ultimately the benthic population will be dominated by the Oligochaeta.

Fish Production

It has been noted that the annual commercial fish production in Lake Winnipeg declined in 1961 and has since remained at levels below the 38-year average (Fig. 3). Overfishing and the use of illegal mesh nets were believed to be the major factors producing the decline (Personal communication, Mr. W. Pollard, Manitoba Department of Mines and Natural Resources). The closure of commercial fishing operations on Lake Winnipeg may permit some recovery of fish stocks.

Hydrographic Data

Since 1961, there has been an increase in Lake Winnipeg water levels terminating in the 1966 maximum (Fig. 4). Increased precipitation and heavy run-off combined to produce high lake levels accompanied by high chemical concentrations (Table 2). Increased nutrient loading produced high benthic densities (Table 3). Benthic densities more than four times greater than the value of $801/m^2$ in September 1966 have been subsequently recorded. While population increases could have occurred, the concentration method cannot be overlooked as a causative agent.

SUMMARY AND CONCLUSIONS

Between March 1962 and September 1969, surveys on the south basin of Lake Winnipeg were conducted twice yearly. The purpose of these surveys was to determine changes within the basin attributable to pollution. Emphasis was directed to the chemical composition of water and densities and composition of the benthos.

The only ionic component showing an increase since 1930 is calcium. Mean concentrations of calcium, iron, sulphate and total solids were higher in 1969 than in 1930. Maximum levels for most chemical components were recorded in 1966 and 1967, corresponding to high lake water levels. The lack of any significant increases in concentrations is believed to be due to low residence time and effective flushing rates.

Since 1930, benthic densities have increased from values of 437 and $578/m^2$ to $776/m^2$ during the 1962-1969 period. The composition of the benthos has shifted from that dominated by Amphipoda and Ephemeroptera in 1930 to the present Chironomidae - Oligochaeta configuration. Approximately 38% of all chironomids and 95% of all oligochaetes, between 1962 and 1969, were composed of species which could be termed pollutional. As enrichment proceeds, the benthos in the south basin of Lake Winnipeg will ultimately be dominated by oligochaetes.

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APPENDIX 1

Ranges in Values for Chemical Data (mg/l), Lake Winnipeg South Basin, March 1962

Physical and Chemical Factors	1962		1963		1964		1965	
	March	September	March	September	March	September	March	September
True color P+ Co units	-	-	-	-	-	-	-	-
Apparent color P+ Co units	20 to 50 (35.6)	-	40 to 70 (52.5)	-	30 to 60 (48.5)	-	-	-
Turbidity	1.0 to 30 (10.8)	-	4.5 to 34 (15.1)	-	4.0 to 32.5 (17.1)	-	6 to 20 (10.3)	-
pH	7.0 to 7.8 (7.3)	7.2 to 8.4 (7.6)	6.7 to 7.6 (7.3)	6.6 to 7.6 (7.2)	6.7 to 8.0 (7.2)	7.2 to 8.1 (7.7)	7.4 to 8.2 (7.8)	7.5 to 7.9 (7.6)
Calcium	16.8 to 94.4 (26.3)	-	16.0 to 40.0 (23.9)	14.4 to 36.8 (28.9)	15.2 to 36.0 (25.2)	14.4 to 60.0 (25.0)	0.8 to 36.0 (28)	22.0 to 32.8 (26.4)
Magnesium	-	-	-	3.9 to 13.6 (10.2)	-	4.9 to 12.2 (9.0)	-	1.7 to 12.6 (9.3)
Sodium	-	-	-	2.5 to 13.5 (8.2)	-	2.0 to 9.5 (6.4)	-	4.5 to 11.5 (7.6)
Iron	0.6 to 3.9 (0.4)	-	0.1 to 1.08 (0.5)	0.3 to 2.3 (0.8)	0.12 to 0.68 (0.34)	1.4 to 3.9 (2.3)	-	0.36 to 1.4 (0.73)
Total hardness	54 to 382 (96.7)	-	54 to 160 (91.3)	52 to 148 (115)	54 to 126 (99.6)	56 to 122 (94.2)	66 to 140 (105.8)	41 to 134 (104)
Phosphate (ortho)	-	-	-	-	-	-	-	0.11 to 0.40 (0.21)
Alkalinity	18 to 158 (34.4)	92 to 226 (144)	40 to 194 (99.8)	44 to 122 (89.5)	40 to 181 (104.4)	48 to 94 (75.3)	52 to 116 (87.7)	40 to 98 (77)
Bicarbonate	-	-		53 to 132 (106)	-	58 to 115 (91.3)	-	40 to 110 (94.3)
Carbonate	-	-	-	0 to 4.8 (1.2)	-	-	-	-
Chloride	2.0 to 20.4 (6.8)	-	2.0 to 13.5 (5.6)	1.0 to 11.0 (6.5)	1.5 to 100 (5.6)	1.2 to 10.0 (5.9)	2.0 to 13.5 (6.7)	4.0 to 11.5 (8.5)
Carbon dioxide	4.6 to 732 (9.5)	4.0 to 44 (13.7)	1.7 to 6.8 (3.2)	-	0 to 20.4 (3.9)	-	-	-
Dissolved oxygen	6.5 to 16.9 (13.7)	8.2 to 11.2 (9.4)	9.6 to 17.1 (13.1)	9.2 to 9.8 (9.5)	3.2 to 17.7 (12.1)	-	8.0 to 17.5 (13.9)	-
B. O. D.	1.0 to 7.0 (3.7)	-	2.9 to 8.4 (4.9)	0.5 to 1.4 (0.9)	0.4 to 6.3 (3.1)	-	2.0 to 4.1 (3.3)	-
Sulphate	-	-	-	8.2 to 40.4 (28.3)	-	11.5 to 37.8 (28.3)	-	28.1 to 49.5 (38.8)
Spec. cond. µ mhos	-	-	-	-	-	-	-	-
Total solids	38 to 650 (161)	-	94 to 284 (56)	106 to 238 (190)	80 to 210 (148)	110 to 286 (214)	102 to 368 (164)	150 to 216 (173)
Suspended solids	-	-	1.5 to 27 (10.4)	-	1.0 to 9.0 (3.9)	-	0 to 37 (14.4)	-
Dissolved solids	-	-	-	82.0 to 309.9 (189)	-	-	-	-
Number of samples			22	24	32	18	39	8

APPENDIX 1 (cont'd)

Ranges in Values for Chemical Data (mg/l), Lake Winnipeg South Basin, March 1962

Physical and Chemical Factors	1966		1967		1968		1969	
	March	September	March	September	March	September	March	September
True color P+ Co units				15 to 25 (19)		15 to 45 (25)	30 to 40 (36)	30 to 100 (42)
Apparent color P+ Co units								
Turbidity		9 to 50 (26)	50	4 to 130 (38)	15 to 55 (31)	5 to 50 (18)	3 to 30 (16)	10 to 320 (35)
pH	7.3 to 7.7 (7.5)	7.9 to 8.0 (8.0)	7.8	7.4 to 8.2 (7.8)	7.98 to 8.11 (8.07)	7.33 to 8.20 (7.83)	7.25 to 7.72 (7.58)	7.65 to 8.99 (7.98)
Calcium	14.4 to 36.8 (22.4)	17.6 to 44.2 (33.7)	59.2	27.2 to 48.0 (36.2)	30.4 [illegible]8.4 (3[illegible])	17.6 to 49.6 (30.0)	27.2 to 31.2 (28.0)	12.8 to 52.0 (29.0)
Magnesium	2.9 to 12.6 (7.0)	10.9 to 18.7 (14.2)	32.1	9.7 to 21.9 (13.8)	7.3 to 12.6 (10.8)	0.28 to 19.4 (9.3)	8.9 to 11.2 (9.9)	2.9 to 22.8 (11.1)
Sodium	2.0 to 13.0 (5.2)	9.5 to 17.8 (13.3)	26.4	8.6 to 18.8 (11.4)	8.3 to 13.0 (10.4)	3.2 to 17.6 (9.7)	9.3 to 12.0 (10.1)	2.7 to 35.3 (12.1)
Iron	0.14 to 0.32 (0.2)	0.11 to 1.8 (0.98)	14.2	0.04 to 1.63 (0.44)	0.58 to 2.7 (1.5)	0.14 to 3.12 (0.59)	0.16 to 1.04 (0.28)	0.20 to 7.0 (1.31)
Total hardness	48 to 144 (85.2)	118 to 188 (151)	280	112 to 240 (147)	110 to 144 (126)	57 to 196 (113)	105 to 122 (110)	48 to 224 (116)
Phosphate (ortho)	0.11 to 0.22 (0.14)	0.34 to 1.3 (0.53)	0.39		0.05 to 0.11 (0.08)	0.0 to 0.36 (0.13)	0.13 to 0.27 (0.17)	0.085 to 0.44 (0.23)
Alkalinity	40 to 106 (63.6)	90 to 139 (113)	182		82 to 120 (98)	44 to 144 (86)	76 to 88 (83)	44 to 168 (92)
Bicarbonate	48.8 to 129 (77.1)	110 to 170 (138)	222		100 to 146 (119)	53 to 176 (105)	93 to 107 (100)	54 to 205 (114)
Carbonate								
Chloride	0 to 12 (4.3)	0.2 to 13.4 (5.4)	19	11.5 to 19.8 (15.7)	9.1 to 13.3 (11.0)	2.5 to 18.0 (8.8)	7.0 to 8.0 (7.3)	1.5 to 22.8 (8.8)
Carbon dioxide					4 to 10 (7.3)		1.9 to 32.4 (7.2)	0.1 to 1.5 (0.3)
Dissolved oxygen	10.2 to 16.5 (14.4)	7.7 to 10.0 (9.0)	10.0 to 16.3 (13.5)	7.3 to 10.6 (8.7)	9.0 to 10.1 (7.1)	9.8 to 10.4 (9.9)	4.5 to 16.0 (12.2)	8.3 to 9.8 (8.7)
B.O.D.								0.4 to 2.3 (1.3)
Sulphate	17 to 48 (28)	16.5 to 58.4 (36.5)	45.3	28.2 to 60.0 (41.8)	28.8 to 41.2 (35.1)	3 to 60 (34)	24.7 to 31.3 (29.4)	3.6 to 8.2 (4.0)
Spec. cond. μ mhos					195 to 290 (229)		235 to 270 (252)	110 to 580 (298)
Total solids	82 to 212 (132)	160 to 280 (211)	642	170 to 346 (206)		9.0 to 336 (180)	150 to 202 (175)	98 to 566 (246)
Suspended solids				4 to 64 (14)		0 to 50 (15)	0 to 52 (10)	6 to 62 (22)
Dissolved solids	74 to 208 (127)	96 to 276 (183)	338	158 to 304 (203)		80 to 290 (160)	146 to 180 (165)	92 to 374 (212)
Number of samples	20	8	1	34	20	32		33

APPENDIX 2

Larval Forms of Chironomidae Collected

Subfamily Tanypodinae	**Reference**
Tanypus sp.	Mason, 1968
Procladius sp.	Roback, 1957
Procladius riparius (Malloch)	–
Procladius culiciformis (Linnaeus)	–
Procladius nr. *adumbratus* (Johannsen)	–
Coelotanypus sp. (Kieffer)	Mason, 1968
**Clinotanypus* sp. (Kieffer)	–
**Zavrelimyia* sp.	–
Pentaneura carnea (Fabricius)	–
Ablabesmyia sp.	–
Subfamily Diamesinae	
**Diamesa* sp.	Mason, 1968
Subfamily Orthocladiinae	
**Cardiocladius* sp. (Kieffer)	Mason, 1968
**Heterotrissocladius* sp.	–
**Nanocladius* sp.	–
**Orthocladius* sp.	–
**Psectrocladius* sp. (Kieffer)	–
**Trichocladius* sp. (Kieffer)	–
Subfamily Chironominae	
Chironomus sp.	Curry, 1962
Chironomus attenuatus (Walker)	–
Chironomus riparius (Meigen)	–
Chironomus plumosus (Linnaeus)	–
Chironomus tentans (Fabricius)	–
Chironomus taxis (Curran)	–
Chironomus staegeri (Lundbeck)	–
Chironomus anthracinus (Zetterstedt)	–
**Chironomus neomodestus* (Malloch)	–
**Chironomus tendipediformis* (Goetghebuer)	–
**Chironomus paganus* (Meigen)	–
**Chironomus hyperboreus* (Staeger)	–
**Chironomus ochreatus* (Townes)	–
Chironomus (*Kiefferulus*)	Mason, 1968
Chironomus (*Einfeldia*)	–
Microtendipes sp.	Curry, 1962
Glyptotendipes sp. (Kieffer)	–
Cryptochironomus digitatus (Malloch)	Roback, 1957
Cryptochironomus falcus (Johannson)	–
Polypedilum (*Tripodura*) *scalaenum* (Schrank)	–
Chironomus (*Cryptochironomus*) *nais*	Mason, 1968
Cryptochironomus (*Harnischia*)	–
Tanytarsus (Vander Wulp) = *Calopsectra* (Kieffer)	–
Harnischia (Kieffer)	Roback, 1957

* *The identity of these genera and species is in doubt. Few specimens were obtained and all were larvae.*

EUTROPHICATION IN SOME LAKES AND COASTAL AREAS IN FINLAND, WITH SPECIAL REFERENCE TO POLYHUMIC LAKES

Pasi O. Lehmusluoto

INTRODUCTION

The dominant feature of the Finnish inshore waters is the fact that the lakes are linked together by short stretches of running water. They form in this way watercourses (Fig. 1). The number of lakes is estimated to be within the range of 55,000 - 75,000. Their total area is 32,000 km^2 and water volume is about 220 km^3 (mean depth about 7 meters). The mean run-off (MQ) is 95 km^3/year. The average color of the water is comparatively high in the whole country, i.e. 91 mg Pt/l. This means that the average humus content of the water is about 13 mg humus (dry weight) per liter (Ryhanen, 1968). The iron content is also comparatively high, and averages 1.1 mg/l. The lake water is normally clear, slightly acid (pH 6.6), and the conductivity is low, under 70 micromhos (Laaksonen, 1970).

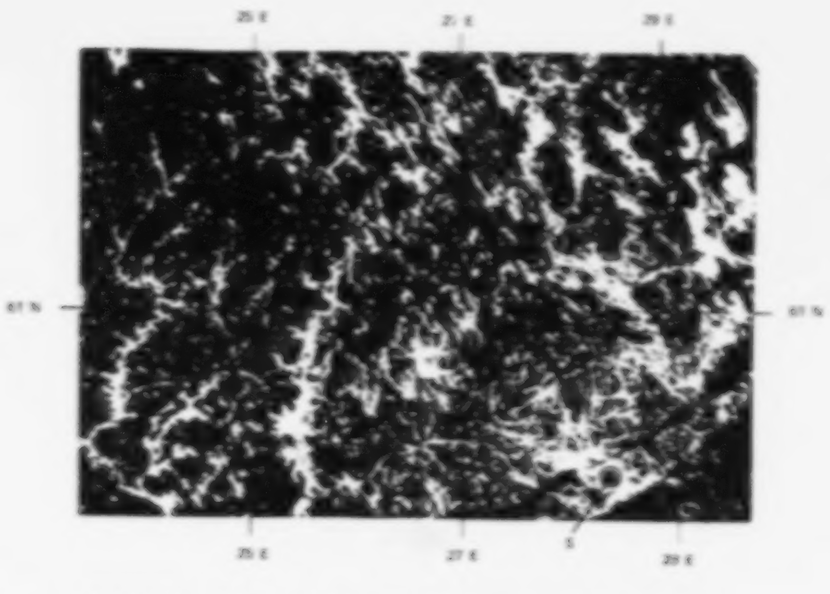

FIGURE 1 A part of the Finnish lake district. S= Lake Saimaa.

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Allotrophic humus, and organic matter produced in autotrophic phytoplankton primary production, especially in connection with eutrophication, have been found to be most important factors in protection of polyhumic lakes in maintaining sufficient oxygen concentration in the water bodies (Ryhanen, 1968). In certain areas carbon rich pulp mill effluents may also play an important role in loading the waters.

The lakes in Finland are normally dimictic. The spring circulation is not, however, always complete in small lakes. The ice period lasts on the lakes for 150 - 250 days, making the conditions in the water more favorable for decomposition than for phytoplankton primary production. In the hypolimnion the oxygen consumption caused by different kinds of oxidation processes (e.g. the decomposition of organic material produced in primary production, the decomposition of pulp mill effluents, and ferrous iron oxidation) may be great, and lead to an oxygen deficit during winter and summer stagnations. This is especially severe during summer stagnations, when the preceding spring circulation has not been complete.

Increase in phytoplankton primary production is perhaps not as important in polyhumic lakes as in the nonhumic lakes. This is due to the "inhibitive" effect of humus to the total daily phytoplankton primary production (Shapiro, 1957). This effect is caused mainly by the strong light absorption of humic material. In algal assay bottle tests in constant light, humus has shown to be slightly biostimulative (Demmerle, 1967). The visible light (E 400 - 700 nm) penetrates polyhumic lakes only 2-3 meters (Fig. 2). Humus alone depresses the trophogenic layer beyond the normal properties of the water. Thus in polyhumic water bodies there is produced daily only a part of that phytoplankton material produced in similar nonhumic waters.

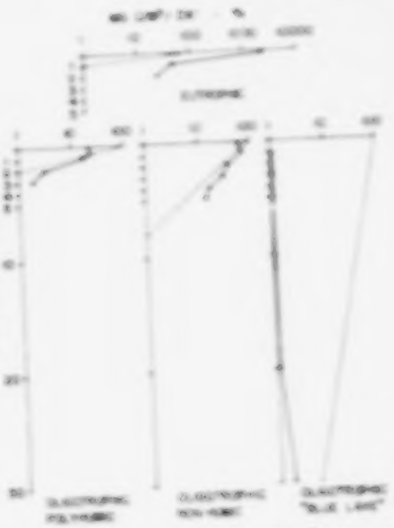

FIGURE 2 **Phytoplankton primary production (mg C/m^3/day, solid line) and visible light extinction (%, dashed line) vs. depth in different kinds of waters in the middle of the growing season. Data for the oligotrophic "blue lake" is from Rodhe et al. (1966). Phytoplankton production and light extinction are in logarithmic scale.**

In some clear water lakes, so called "blue waters", where the optical purity of the water is great (Fig. 2), the effect of ultraviolet radiation can cause a reverse vertical distribution of phytoplankton primary production (Rodhe et al., 1966). Ultraviolet radiation depresses this production below its potential capacity because phytoplankton production can proceed only in the layers where the influence of ultraviolet radiation is negligible and there is still visible light illumination. In these lakes this layer is often beyond the optimum visible light layer in the lower epilimnion (Rodhe et al., 1966).

Normally in nonhumic and nonturbid lakes increasing phytoplankton mass causes the trophogenic layer to be depressed, and the maximum phytoplankton primary production per cubic meter (in optimum visible light layer) is moved closer to the water surface. In polyhumic and very turbid lakes this does not happen because of the already thin trophogenic layer (Fig. 2).

In Finnish lakes, as in some mountain lakes of Austria (Pechlaner, 1964), high phytoplankton primary production beneath the ice during spring time normally cannot be found. The phytoplankton primary production is at that time barely measurable. The lack of an under-ice spring bloom may be partly due to the relatively fast ice melting, as the ice does not remain on the water long enough after the snow has melted to allow sufficient light for intensive phytoplankton production. Low rates of primary production can be measured under ice normally during early winter before the ice is covered with snow (Pechlaner, 1964).

In Alaska almost half of the annual phytoplankton primary production may occur beneath the ice sheet in spring time, e.g. Lake Schrader and Lake Peters (Hobbie, 1964). The highest phytoplankton production for the whole year in Lake Schrader was also found in spring under the ice. Rodhe et al. (1966) measured considerable under-ice phytoplankton production in Swedish Lapland. The high under-ice production in the two Alaskan lakes in spring time was a result of the relative lack of turbulence in the water beneath the ice. The algae remained in the euphotic zone continuously during ice period. After the ice had left and the lakes began to circulate, the phytoplankton primary production dropped sharply. In Finnish lakes the phytoplankton primary production usually increases sharply during this period. At this time of year the algae can stay in the euphotic zone only a small part of their lives, as in polyhumic lakes during summer time. These algae also appear to be adapted to low light intensities (Hobbie, 1964).

The growing season is short in Finland with daylight in June averaging 18 hours, August 16 hours, but in October only 10 hours. The ice period normally lasts from November to April.

The vernal diatom bloom is dependent on the length of the day and on the illumination during the day (Gran, 1929). This bloom can be normally found in the Finnish waters from the end of April to the end of May, when the solar radiation exceeds 2.5 - 3.5 kcal/m^2/min. The vernal diatom bloom could possibly be found when the solar radiation exceeds 1.5 kcal/m^2/min (Smayda, 1959), but the bloom is hindered at that time by the ice cover (Lehmusluoto, 1968).

During summer time in polyhumic lakes, the lack of light and in some Alaskan and Swedish mountain lakes, the overwhelming light (Hobbie, 1964, Rodhe et al., 1966) seems to limit the total daily primary production per square meter of the water, but the nutrients may limit the maximum primary production of a cubic meter of water (Hobbie, 1964).

When discussing primary production, one cannot help dealing with the lake typology. Nowadays it has been suggested that primary production could be the basis for the trophic classification of waters (Rodhe, 1958, 1969, Findenegg, 1964). There are many parameters of primary production that could be used: 1) the shape of the primary production curve (Findenegg, 1964), 2) primary production per square unit (diurnal, annual) (Rodhe, 1958, Findenegg, 1964, Hübel, 1968), 3) maximum primary production per unit volume (Rodhe, 1958, Lehmusluoto, 1969) 4) so called V/O-quotient as suggested by Rodhe (1958).

I would like to suggest that the mean maximum primary production rate per unit volume in the water column during the growing season measured *in situ*, or in constant light, could be used as an index for many kinds of water bodies. This would give a relative but objective index for the phototrophic level of the water. It would not reflect the total amount of organic material built up in the whole water column (which can be estimated by the light extinction curve), but it might be informative about the degree of eutrophication of the water. No suggestions for the designation of oligotrophic or eutrophic waters are presented, because the data available is insufficient.

When talking about lake typology, especially of polyhumic waters, it may be appropriate to use phytoplankton primary production only as a measure of the autotrophic state. The direct comparison of trophic states of humic and nonhumic lakes is impossible, as the classification of polyhumic lakes on the basis of phytoplankton production cannot give an objective result (Ryhänen, 1968). Ohle (1940, 1956) has suggested that also the heterotrophic state should be taken into account.

EXAMPLES OF EUTROPHICATION

Experiments with Polyhumic Water

In order to get some information about the role of nitrate nitrogen and phosphate phosphorus in the eutrophication process of polyhumic waters, a series of preliminary tests were made by algal assay bottle tests (Bringmann and Kühn, 1956; Skulberg, 1964), with water from the polyhumic Lake Hakojärvi (61° 15' N, 25° 12' E). The color of the water was about 150 mg Pt/l and the iron content averaged in the whole water column (maximum depth 15 meters) about 0.5 mg/l. The ice period lasts on this lake about 180 days, from November to the end of April. Spring circulation was complete during 1966 - 1970 only twice.

Results showed that nitrogen was the primary limiting nutrient. Nitrogen was also a limiting nutrient for bacterial growth in this water (Sederholm, 1969). When nitrogen and phosphorus were added together a large algal growth occurred (Fig. 3).

A series of experiments on phytoplankton primary production during 1968-1969 using 7 *in situ* plastic test cells were conducted. The cells were 10 meters deep, 1.2 meters in diameter, and the top was open (Goldman, 1962). The water volume was about 12 m^3. Only the results of the test cells of 1968 are dealt with in this paper, as the results of 1969 were quite similar in nature.

Nitrate nitrogen and phosphate phosphorus were added to the test cells according to the preliminary tests by algal assay in bottles. Nutrient additions varied from 0.05 to 1.0 mg/l nitrogen and 0.005 - 0.1 mg/l phosphorus. The nutrient additions to the different test cells are shown in Table 1.

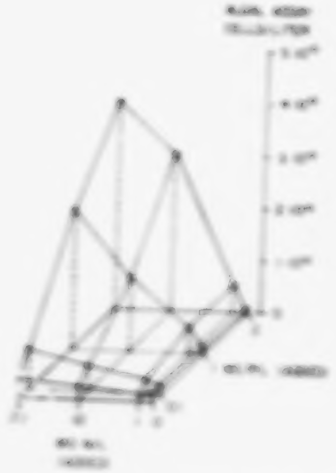

FIGURE 3 The influence of different nitrogen (nitrate) and phosphorus (phosphate) additions in polyhumic lake water (Lake Hakojärvi) as measured by algal assay in bottles. Test alga was *Chlorella* sp.

TABLE 1

Nutrient Additions into the Test Cells

Cell No.	1	2	3	4	5	6	7
NO_3-N, mg/l	-	1.0	-	1.0	0.5	0.1	0.05
PO_4-P, mg/l	-	-	0.1	0.1	0.05	0.01	0.005

Phytoplankton primary production, which was measured by the carbon-14 method (Steemann Nielsen, 1952), was made eleven times during the test period, (17.7. – 8.10. 1968). The test period was about 2/3 of the whole growing season. The background production (Cell No. 1) was 1.58 g C/m²/test period (average for test period 20.2 mg C/m³ (max.)/day). In Lake Hakojärvi in 1966-1969 the phytoplankton primary production averaged 2.66 g C/m² (growing season (average for test period 17.7 mg C/m³ (max.)/day). There were not any large changes in phytoplankton production in the test cell, where only nitrogen had been added, but phosphorus addition alone caused a slight increase (Table 2). Phosphorus did not alone cause significant eutrophication, i.e. phosphate-eutrophication (Thomas, 1969). In the cells where both nitrogen and phosphorus were added the increase in phytoplankton production was greater with greater additions of nutrients. The highest value, 8.16 g C/m²/test period (154.2 mg C/m³ (max)/day in test period), was in the cell containing the greatest nutrient addition (1.0 mg N/l and 0.1 mg P/l).

TABLE 2

Phytoplankton Primary Production g C/m^2/test period and mg C/m^3 (max.)/day on the Average in the Test Period For Nutrient Additions See Table 1

Cell No.	1	2	3	4	5	6	7
g C/m^2/test period	1.56	1.32	1.42	6.16	3.48	2.80	1.92
mg C/m^3 (max.)/day*	29.2	23.5	44.1	154.2	65.3	49.6	51.5

*Average for the growing season (eleven measurements).

The oxygen stratification in the different test cells did not show any great variations. In the epilimnion at a depth of 1 meter each cell contained a range of 8.3 to 8.9 mg O_2/l, and at 5 meters depth from 6.8 to 7.1 mg O_2/l, in the hypolimnion at 10 meters depth between 5.7 to 6.2 mg O_2/l. The highest oxygen values in the epilimnion were normally found in the cells where both nitrogen and phosphorus had been added. However, the lowest values in hypolimnion were not found in these cells. Nutrient additions in these experiments apparently did not stimulate oxygen consumption in the hypolimnion.

The uptake of radioactive glucose by heterotrophic bacteria (Wright and Hobbie, 1965) was also stimulated by nitrogen and phosphorus. In the test cell where the greatest nitrogen and phosphorus addition was made, primary production increased about five times, and the uptake of radioactive glucose increased about ten times (Lappalainen, 1970). The bulk of heterotrophic bacteria which caused the uptake of radioactive glucose may have been dependent on the algal mass produced, because the increase of glucose uptake occurred only in the upper epilimnion where primary production was at a maximum (Fondén, 1969). The same nutrient addition was present both in the lower epilimnion and in hypolimnion.

Although it seems that the humus in the water did not in these experiments form so serious a problem in maintaining sufficient oxygen concentration in hypolimnion as the organic material built up by the phytoplankton primary production, humus may be important in the total production of polyhumic waters. The zooplankton and fish production in the Finnish polyhumic lakes seem to be greater than the phytoplankton primary production can yield (Ryhänen, 1968). It is proposed that humus as such, or transformed to bacterial biomass, may serve as food for the zooplankton (Järnefelt, 1956), and thus lead to a relatively high fish production. It has been shown that the bacterial numbers in humic waters are normally comparable to the numbers in eutrophic lakes (McCoy and Sarles, 1969).

On March 22, 1970 the entire Lake Hakojärvi was fertilized by adding 750,000 m^3 phosphorus and nitrogen to bring the lake water concentration up by 1 mg N/l and 0.1 mg P/l. Results from this experiment are not yet available.

The above experiments show the role of nitrogen and phosphorus in the eutrophication process in the water of Lake Hakojärvi. In order to get some information about eutrophication effects of domestic sewage and pulp mill effluents, which are typical pollutants in Finland, some data is presented in the following from Lake Saimaa (Fig. 1) and two coastal areas in the Gulf of Finland.

Lake Water

In Lake Saimaa, which is one of the largest lakes in Finland, phytoplankton production was studied in two areas during 1968. The first area in northern parts of Lake Saimaa was being polluted by about 12,000 m^3/day of biologically purified domestic sewage. The second area was in the south, and was polluted by 260,000 m^3/day of mechanically purified pulp mill waste. The effluent consists of 44% sulphite and 56% sulphate liquor (Lehmusluoto and Heinonen, 1970). Preliminary tests were made by using algal assay bottle tests mentioned earlier.

Domestic sewage biostimulated the algae growth in every concentration, but pulp mill effluents were almost lethal to algae at 10% effluent concentrations. In 1% and 0.1% effluent concentrations, algae growth was slightly stimulated (Fig. 4).

Primary production measurements from the lake water were made six times during the growing season in constant light (5,000 lux) to eliminate the diurnal changes in illumination.

Domestic sewage caused increased primary production in the recipient. Phytoplankton production, without any depression near the sewage outfall, decreased, as did the oxygen consumption in hypolimnion with distance. In the hypolimnion there was no oxygen deficit in any region. The greatest values of phytoplankton production (294.1 and 299.5 mg C/m^3/day) were observed near the outfall. At a distance of about 24 km from the outfall, the primary production was 48.1 mg C/m^3/day (Table 3). This value can be used as an index for unpolluted water in that region.

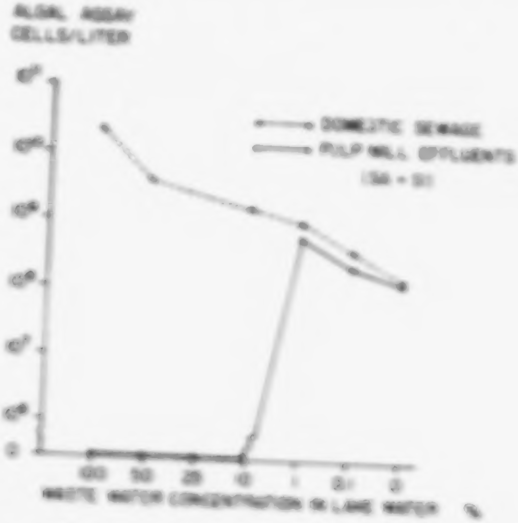

FIGURE 4 The influence of domestic sewage and pulp mill effluents (sulphate, sa and sulphite, si liquor) in lake water (Lake Saimaa) as measured by algal assay in bottles. Test alga was *Ankistrodesmus* sp.

TABLE 3

Phytoplankton Primary Production mg C/m³/day in Northern Parts of Lake Saimaa as Mean Values of Six Measurements During the Growing Season

The Recipient is Influenced by Domestic Sewage

Distance from Outfall, km	2	4	8	12	24
mg C/m³/day	294.1	290.5	239.7	170.4	48.1

The pulp mill effluents seemed at first to hinder phytoplankton primary production, but caused a pronounced eutrophication (752.9 mg C/m³/day) about 9 km from the outfall. At a distance of about 22 km, phytoplankton production was 175.2 mg C/m³/day (Table 4), and the water was still slightly eutrophic.

It is shown that both domestic sewage and pulp mill effluents did cause eutrophication of the recipient. Sewage did not have an inhibitive effect as did the effluents. The maximum eutrophication caused by the pulp mill effluents was more intensive than that caused by the domestic sewage, but it occurred far from the discharge area.

TABLE 4

Phytoplankton Primary Production mg C/m³/day in Southern Parts of Lake Saimaa as Mean Values of Six Measurements During the Growing Season

The Recipient is Influenced by Domestic Sewage

Distance from Outfall, km	-4	2	4	9	13	22
mg C/m³/day	93.1	177.9	285.6	752.9	549.8	175.2

Coastal Water

Eutrophication is not only a problem of lakes but also of some coastal areas. Coastal waters in Finland are brackish waters, partly meso- and oligohaline. Problems are concentrated usually near the cities on the coast.

Helsinki, the capital of Finland, initiated in 1965 a research program on the pollution of the Baltic around the city. This was initiated because the archipelago near the city is an important recreation area for the citizens during the summer time and the quality of the water was decreasing with increasing eutrophication. This was mainly due to blue-green algae blooms in the later summer.

The City of Helsinki was discharging its mechanically and biologically purified waste waters into the nearby bays. These purification procedures seemed inadequate to control the eutrophication in this coastal area.

One part of the research program consisted of phytoplankton primary production studies. In this paper the results uf one of the fjord systems of the year 1967 will be briefly dealt with.

Phytoplankton production, mg C/m^3 (max.)/day, in the middle of the growing season at 1 km from the outfall was about 30 timer higher than that at 12 km distance in the unpolluted area outside Helsinki (Fig. 5). On the average during the growing season it was about 20 times higher (Table 5). Fluctuations in primary production per square meter became irregular with increasing eutrophication (Fig. 6). Annual primary production in the innermost bay was 174 g C/m^2/year, i.e. over 5 times higher than in the unpolluted areas in the Gulf of Finland where it was 30 g C/m^2/year (Table 5).

TABLE 5

Primary Production g C/m^2/Growing Season and mg C/m^3 (max.)/day on the Average in the Growing Season in a Fjord System in the Gulf of Finland off the Coast of Helsinki in 1967

Distance from Outfall, km	1	3	4	5	8	12
g C/m^2/growing season	174	132	96	51	48	30
mg C/m^3 (max.)/day*	1092.6	950.8	533.9	133.9	99.6	58.9

*Average for the growing season (eleven measurements).

According to Jonasson (1969), in eutrophic waters phytoplankton primary production acts in the following way:

- Primary production (per cubic meter) increases very sharply
- Fluctuations in primary production (per square meter) becomes irregular and
- Annual primary production (per square meter) may increase sharply.

In coastal areas pulp mill effluents have a quite similar influence as in the lakes. Two series of experiments of phytoplankton production in constant light, made in 1967 near the City of Kotka in the Gulf of Finland, show this (Table 6). The phytoplankton primary production did increase to a distance of 10-15 km from the outfall. In June, phytoplankton primary production did increase to a distance of 10-15 km from the outfall. In June phytoplankton production was low and it did not increase at 20 km distance from the outfall. In August eutrophication was obvious and it reached beyond 20 km from the outfall.

CONCLUSIONS

Eutrophication of waters is a natural process, but as seen above, it can be greatly accelerated by man. Therefore, it has become a recent problem in water protection. The increases in population, industry and agricultural activities, for example, introduce excess nutrients and other pollutants into the waters. Nutrients may accelerate the development of eutrophication.

TABLE 6

Primary Production near the City of Kotka in the Gulf of Finland in 1967

Data from Lehmusluoto (1967)

Distance from Outfall, km	2	6	10	15	20
June, mg C/m^3/day	14.1	56.7	78.5	59.8	39.7
August, mg C/m^3/day	15.6	63.6	115.9	219.3	175.9

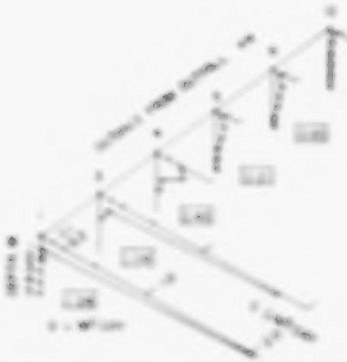

FIGURE 5 Phytoplankton primary production off the coast of Helsinki in the Gulf of Finland at different stations on July 18, 1967. The daily phytoplankton production is given in g C/m^2/day.

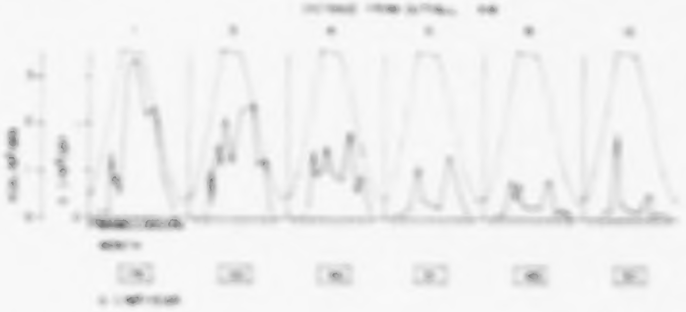

FIGURE 6 Phytoplankton primary production g C/m^2/day (solid line) at different stations off the coast of Helsinki in the Gulf of Finland in 1967. Total solar radiation (10^7 %) × kcal/m^2/mon (dashed line) (Simojoki, 1958). The annual phytoplankton production is given in g C/m^2.

Phytoplankton primary production – as used as a measure of eutrophication – reflects the ecological factors and algal succession. The most important factors are illumination, temperature and nutrient concentration of the water. The annual phytoplankton production (intensity of eutrophication) is almost wholly dependent on the nutrient concentration, and on the availability of these nutrients. The influence of the other factors, despite annual fluctuations, may be almost constant.

Domestic sewage and pulp mill effluents are important nutrient sources causing intense eutrophication. Sewage is, thus, not the only type of waste water to cause eutrophication. Pulp mill effluents may also play a great part in the process, although their first influence seems to be to hinder phytoplankton production. At some distance from the outfall, when the effluents have been diluted, phytoplankton production can proceed and the water eutrophicate.

The most important nutrients are nitrogen and phosphorus. Other substances, such as vitamins and trace elements, must also be available. Many other growth stimulators are found in sewage.

It is important to consider all the possible efforts to reduce nutrient input to receiving waters in any form in order to avoid overeutrophication of waters.

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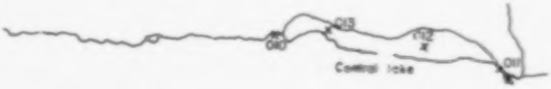
THE RECOVERY PROCESS OF A LAKE WHICH RECEIVED WASTEWATER FROM AN ORE DRESSING PLANT

Bengt Ahling

A question, on which interest is currently focusing in the problem-complex of industrial pollution, is what happens to a recipient of industrial waste water if the industry closes down or improves its waste water plant so that the addition of polluted water appreciably diminishes.

The question is of very great importance from the standpoint to be adopted in reference to measures for the restoration of polluted lakes. In those cases in which a recipient cannot, within a reasonable time, purify itself to the point to which its water can be used for the different purposes that may be considered desirable, it may be necessary to take steps to restore the lake.



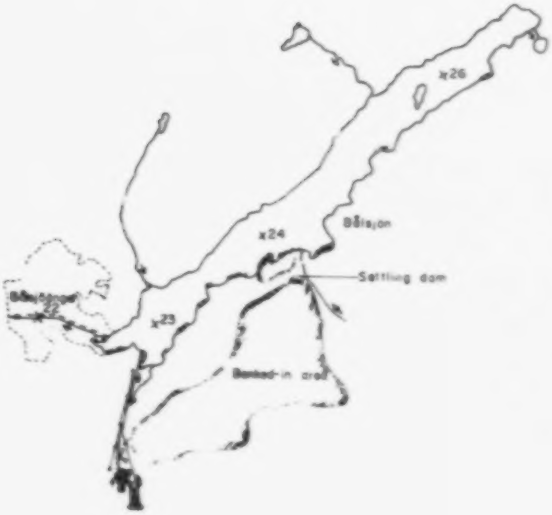


FIGURE 1 Lake Bålsjon

One of the lakes that has been investigated in order to study the recovery process is Lake Bål (Bålsjon) in Central Sweden. The area of Bålsjon is 0.30 km^2, and the mean and maximum depths are 5 and 11 meters respectively. The catchment area is rather small, only 5.5 km^2, and consists mostly of marshes. Up to the autumn of 1967 Lake Bål was receiving the waste water from a dressing plant for magnetite and hematite. The dressing plant operated with gravity and wet-magnetic concentration and therefore had to use large quantities of water. As the smallest particles were very hard to separate off with the methods used, they followed with the water through the dressing plant.

The waste water was pumped to a banked-in area for sedimentation (Fig. 1). From here the still turbid water was conducted via a settling dam to Bålsjon. This made the water in Bålsjon so turbid that the Secchi disk transparency was reduced to only a few centimeters. Owing to this turbidity, light could not penetrate to any appreciable depth, but was reflected or absorbed in the topmost layer of water. In Bålsjon there was constant sedimentation of the fine-grained particles. This meant that any organic substance eventually formed in or added to the lake was immediately embedded in the sediment, which made Bålsjon a very sterile milieu for many organisms.

Before the plant closed down, Bålsjon and a similar lake used as a control lake were investigated for a couple of years, so that there was comparative material when it came to a study of the recovery of the lake. In order to be able to give a clear picture of the recovery it may be as well to give an account of the situation prevailing before the plant closed down.

QUALITY OF THE WATER WHEN THE DRESSING PLANT WAS OPERATING

Chemico-physical Conditions

Turbidity, Color and Secchi Disk Transparency

The most striking effect of the discharge from the dressing plant was the marked turbidity of the recipient. The heamtite sludge that was not utilized imparted a distinct reddish color to Bålsjon. Through the discharge this coloring was much intensified. The control lake had a color value of about 30 mg Pt/l, while values as high as 650 mg Pt/l were measured in Bålsjon.

Owing to the turbidity, the Secchi disk transparency was reduced to only a few centimeters.

pH and Conductivity

The waste water had a pH between 8.2 and 8.3 in the sedimentation basins. In Bålsjon, after dilution, pH was between 7.8 and 8.1 in the spring and summer samples respectively. Owing to the abundance of unbound ions in the waste water, the specific conductivity was considerably increased. The value in Bålsjon was about seven times that in the control lake.

Iron Content

Due to the waste sludge, the iron content of the water was for the most part very high, as the iron content of the particles was about 15%. The highest content measured in Bålsjon was 10 mg Fe/l, which coincided with a markedly reddish coloring and turbidity. Analysis of the filtered and unfiltered samples showed that most of the iron occurred as suspended material. A not inconsiderable part (25%), however, occurred in such a form that it passed a fine filter paper.

Content of Oxygen

The oxygen content was always high in Bålsjön, showing values of about 90% saturation. The waste from the dressing plant thus did not affect the oxygen content negatively.

Consumption of Permanganate

The consumption of potassium permanganate in the water was determined. The result showed that very small amounts of organic chemically oxidizable substance existed in the water. The values measured were about half of those for the control lake, or 10 mg $KMnO_4$/l.

Biological Conditions

The Bålsjön water had a consistency of extremely fine-grained red sediment. This is unsuitable as a substrate for more or less sessile living lake bed organisms. Bålsjön was therefore characterized by the absence of such species as well as the greater part of the submerged plants. In the samples taken from the lake bed, only a few odd specimens were found of *Chaoborus*, which swims freely in the water near the bed, and of crustaceans of copepod type. In the northern part of the lake there were indications of organic flocculi, and in connection with these a few protozoons, Flagellata, and diatoms like *Synedra*.

Owing to the marked turbidity in Bålsjön, the light could not penetrate to any appreciable depth, so that autotrophic organisms were able to exist only in a very thin surface layer. This implied a marked diminution of the lake's total production.

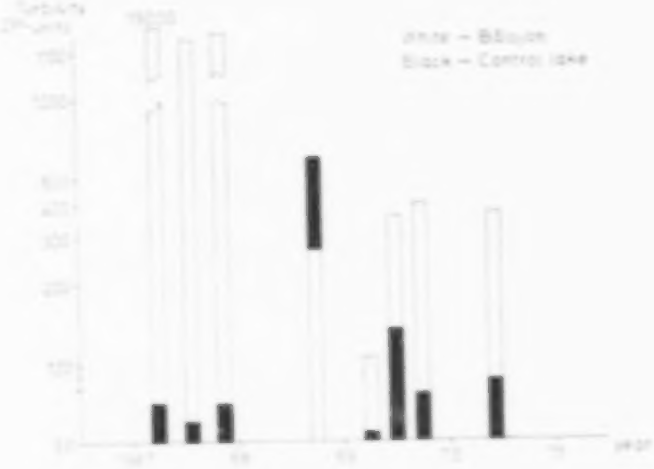


FIGURE 2 Lake Bålsjön: Turbidity vs. Time

This meager production was especially noticeable in the result of the zooplankton counts, where no individuals at all were found. There were, on the other hand, small specimens of pike (*Esox lucius*), roach (*Rutilus rutilus*), perch (*Perca fluviatilis*) and freshwater crayfish (*Potamobius astacus*).

THE RECOVERY PROCESS IN BÅLSJÖN AFTER THE CLOSING DOWN OF THE DRESSING PLANT IN NOVEMBER 1967

Changes In The Physic-chemical Factors

Turbidity, Color, Iron Content and Secchi Disk Transparency

When the discharge of waste water stopped, the particles existing in the water settled. This resulted in reduced turbidity (Fig. 2) and increased Secchi disk transparency (Fig. 3). The lake, which earlier had had a marked red coloring, began to assume a considerably more normal hue. This is also seen in the color measurements in Fig. 4, which show that the color values in Bålsjön were beginning more and more to agree with those in the control lake.

In connection with sedimentation of suspended particles, the iron content was of course reduced to a range of values corresponding with that in the control lake (Fig. 5).

The variations now occurring in turbidity and in the Secchi disk transparency are due to the addition of particles from the surrounding embankment of waste. After a spring flood or heavy rainfall, this addition is noticed as a transient red coloring of Bålsjön.

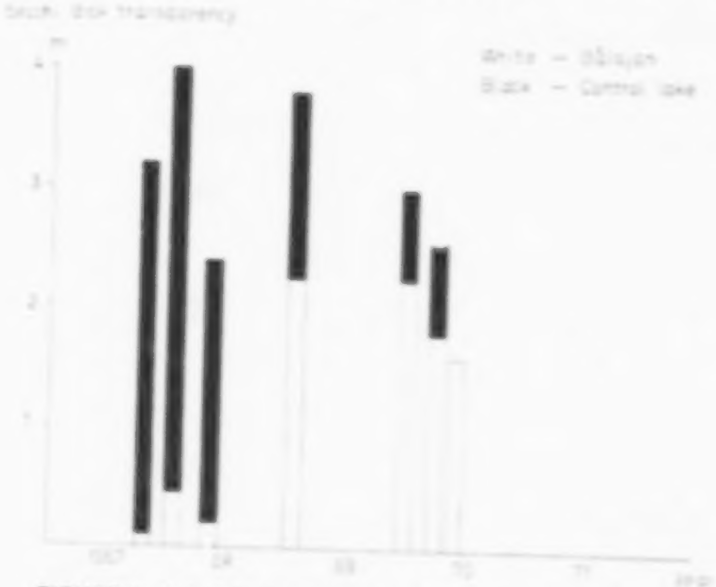


FIGURE 3 Lake Bålsjön: Secchi disk transparency vs. Time

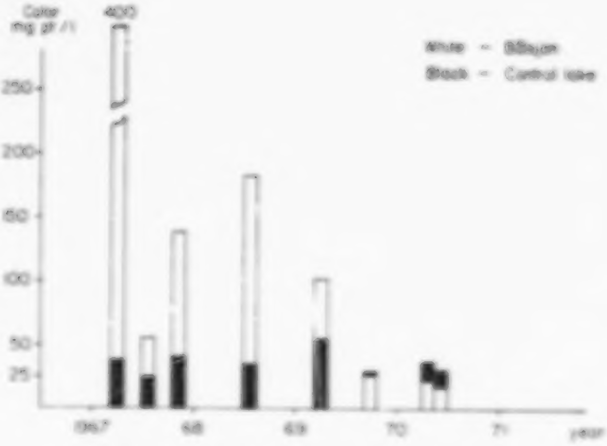


FIGURE 4 Lake Bålsjön: Color vs. Time

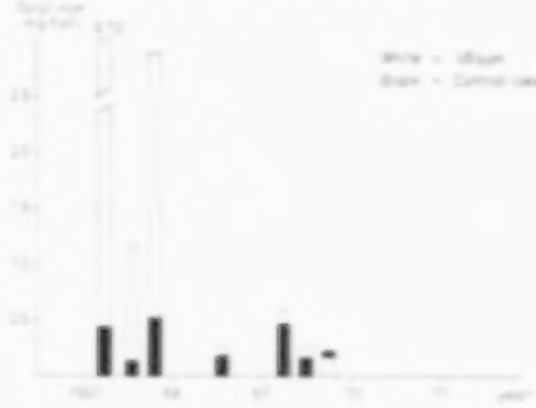


FIGURE 5 Lake Bålsjön: Iron content vs. Time

pH and Conductivity

A certain reduction of pH was observable in the spring samples, while the autumn samples even showed an increase. This may be interpreted to mean that the concentration of the pH-increasing substances had been reduced, which led in turn to a reduction of pH. Furthermore, the primary production increased, which gave increased pH during the production period.

The big difference earlier noted between the specific conductivity of Bålsjön and the control lake respectively began gradually to lessen. (Fig. 6). The slowness of this lessening was due to the very slight water turnover in the lake.

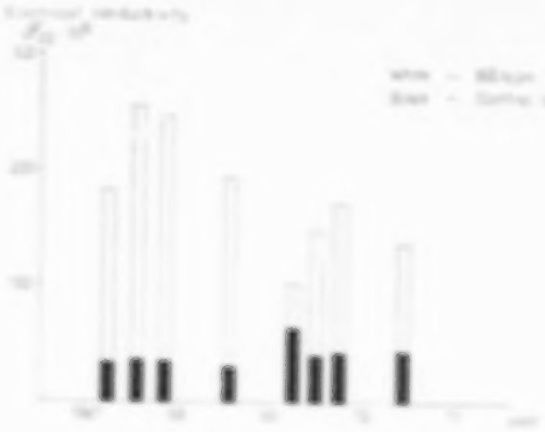


FIGURE 6   Lake Bälgan   Electrical conductivity vs. Time

Oxygen Content and Consumption of Permanganate

After the closing down of the dressing plant the values for the oxygen content were still high. In some samples of water from the bottom of the lake, however, an oxygen deficit down to 12% saturation was observable. This indicates that organic substance may have been added or formed in the lake in such quantities as to have had an effect on the oxygen economy of the lake.

The consumption of permanganate had increased to such an extent in Bälgan that it coincided with that in the control lake, (Fig. 7) which implied that the quantity of organic substances in the

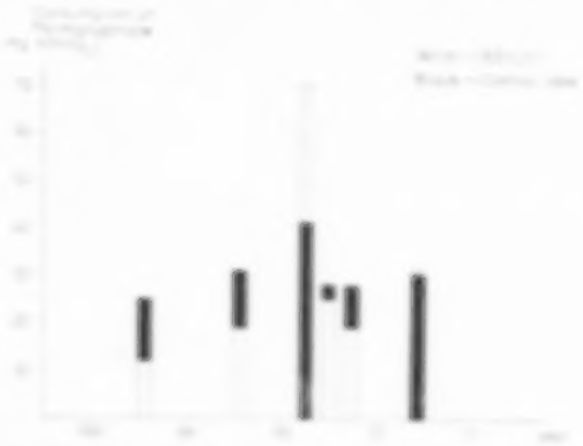


FIGURE 7   Lake Bälgan   Permanganate consumption vs. Time

water had increased. This increase was an indication that the primary production in the lake had increased, or that permanganate-consuming substance had been added to the lake.

Nitrogen and Phosphorus

Nitrogen and phosphorus show higher values as in the control lake. There has not yet, however, been any change in the values.

Changes to The Biological Conditions

Phytoplankton

After the disappearance of turbidity the occurrence of phytoplankton markedly increased (Fig. 8). This applied especially to the diatoms, where during the autumn and winter *Cyclotella comensis* and to a certain extent also *Synedra acus* showed high values for the number of individuals (on 4 November 1969, for instance, there were 2 million *Cyclotella* per liter). In the spring and summer the Chrysomonad *Rhodomonas lacustris* and the earlier completely absent *Dinobryon divergens* occurred in considerable quantities.

Apart from this increased number of individuals, the number of species, especially of Chlorophyta, increased (Fig. 9). Newly added species were e.g. *Elakatothrix gelatinosa* and *Gloeocystis*. The new species in Bålsjön were still of very slight importance for the total production of the lake. From the general picture of plankton it emerged that there was a relatively large number of individuals belonging to a small number of species (Fig. 10); this gave the picture of an extreme milieu. However, the increasing number was an indication that the milieu was becoming less extreme.

If we compare the composition of species in Bålsjön with that in the control lake (Fig. 11) we find a considerably greater variety in the latter, with a larger number of species represented.

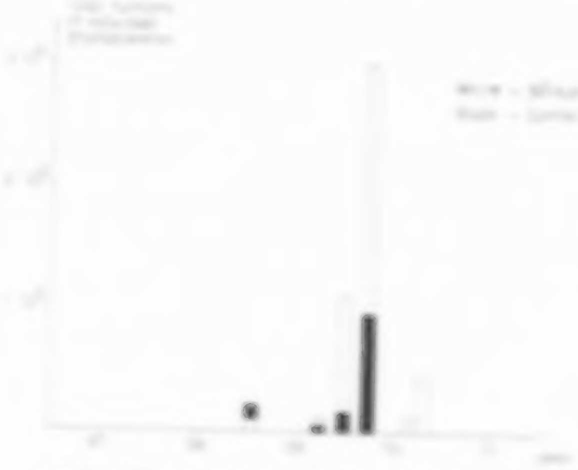


FIGURE 8 Lake Bålsjön: Phytoplankton vs. Time

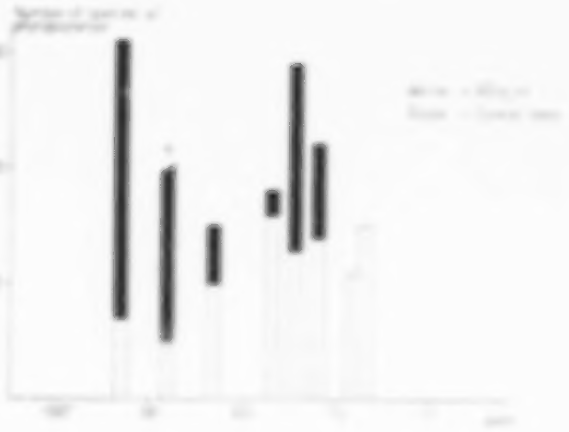


FIGURE 9   Lake Bäljan: Phytoplankton species vs. Time

Zooplankton

Two years after the closing down of the dressing plant Bäljan showed a zooplankton composition almost identical with that in the control lake. The differences referred exclusively to the number of individuals. Thus the number of Ciliata in the control lake was about 10 times greater than in Bäljan, and other zooplankton were 5 times more numerous. Although Bäljan still showed a paucity in the number of individuals as compared with the control lake, the zooplankton existing there implied a marked increase compared with the earlier total absence of these organisms.

Among the zooplankton occurring, the predominant species were the rotifers *Polyarthra*, *Gastropus stylifer* and *Keratella cochlearis*. Other zooplankton species of Cladocera and copepods occurred in Bäljan only as isolated individuals.

Lake Bed Fauna

When the rain of particles over the sediment surface stopped, the fauna normally living on the lake bed could once more start colonizing it. During the investigations carried out the year after the closing down of the dressing plant, it was possible to observe only very slight changes. One only found scattered specimens of Ephemeridae and Chironomidae in the lake bed samples taken.

It was not until one year later, during the summer and autumn of 1969, that it was possible to observe that a community of organisms was in the process of being built up. Ephemeridae, in 1970, were of general occurrence. It was, however, only *Ephemera vulgata* that occurred, and this gave an impression of the instability of the system.

The groups of lake bed organisms most common in Bäljan two years after the closing down of the dressing plant were Chironomidae and Ceratopogonidae. Of the Chironomidae, *Chironomus plumosus* were abundantly represented, but large numbers of organisms from the subfamily

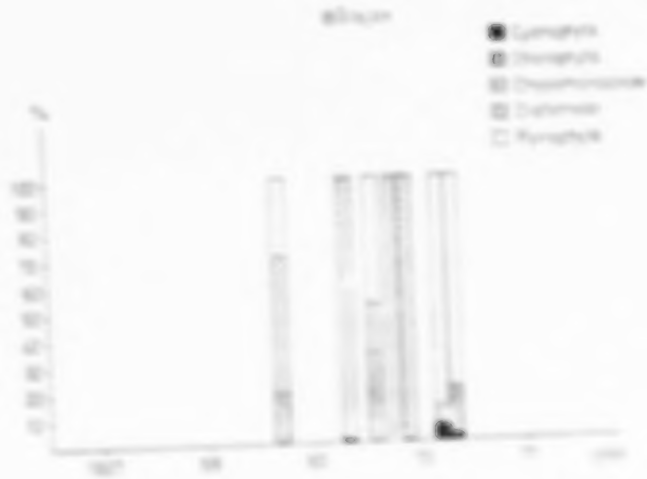


FIGURE 10 Lake Bõhjan: Percent of species vs. Time

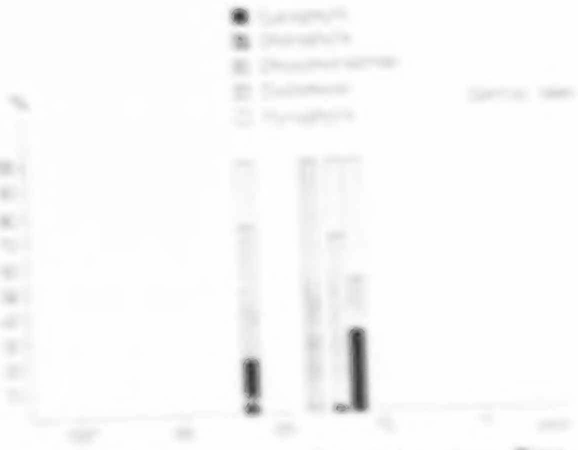


FIGURE 11 Control Lake: Percent of species vs. Time

Tanypodinae also occurred. Of the Ceratopogonidae, *Palpomyia* was the animal occurring most generally in the whole system even if a smaller number of *Chaoborus* also occurred. It was possible to find a greater density of organisms if one went to the northern part of the lake. This was the part furthest from the region where the waste was introduced into Bõhjan, so it thus got the least sedimentation of particles over the sediment surface. It was also possible to find a considerably more modified community of organisms in the discharge canal in the south end of the lake, where the water had little depth and the surrounding reeds contributed organic material which promoted the colonization of the lake bed organisms. Apart from the already mentioned organisms are found in this canal *Herpobdella*, *Asellus* and large quantities of mites.

Owing to the small amount of organic material on the bottom, the lake bed organisms had very poor protection against predacity. This predacity must have certainly been a strongly reducing factor as regards these organisms. Analysis of the gastric contents of fish from Bälsjön also showed the organisms mentioned in the foregoing.

Other Changes

Apart from the changes mentioned in the community of organisms, it was possible to make a number of studies through visual examination of the lake. These changes were very difficult to assess quantitatively, and the observations thus made were, accordingly, very subjective.

On the lake shores one could see great swarms of the ostracod *Notodromas monacha*. This organism commonly occurs in stagnant little lakes during the summer. It is, however, extremely unusual to find such mass development as was here.

At the edge of the shore at the populations of *Phragmites* that existed, it was possible to observe large shoals of one-year-old perch and a large number of pike. Such shoals often occurred near shoals of *Notodromas*, which indicates that the ostracod may be an object of nourishment for the small fry among the fish.

The first traces of *Periphyton* began to appear on the lake bed. These were the thread-formed algae *Zygnema* and *Spirogyra* which had begun to spread to the regions where the light was earlier too weak for these autotrophic algae.

In the cores of sediment collected in Bälsjön in 1969 it was possible to find traces of organic material. This was composed of detritus and vegetable fiber that was presumably only partly produced in the lake and had otherwise come from the soil surrounding the lake. Besides this material there were also shells and remains of diatoms and crustacea.

This study of lake recovery reached one of the most interesting stages in 1970. The first part of the recovery referred chiefly to the physico-chemical factors and was a relatively simple process which was mainly a question of sedimentation rates. The most important question for this process in the near future is how rapidly the embankments of waste along the shores can be bound by vegetation. This is important for the prevention of inorganic material being washed out into the lake by rain and melt water.

The second stage, which referred chiefly to the building up of a balanced biosystem, is a good deal more difficult to forecast. The results hitherto obtained show that the number of species in the system is on the increase. They also show that certain species very easily begin to become predominant with a mass development. This is presumably a tendency that will continue until an organic layer has been formed above the inorganic sediment. This organic layer is essential to give the different organisms protection against predacity. When they have this protection the fluctuations due to this predacity will be reduced.

DEPLETION OF OXYGEN BY MICROORGANISMS IN ALASKAN RIVERS AT LOW TEMPERATURES

Ronald C. Gordon

INTRODUCTION

Several arctic (Lotspeich, unpubl; Schallock, unpubl) and subarctic (Frey, 1969; Gordon, unpubl; Mueller, unpubl; Schallock, unpubl) rivers in Alaska (Alaskan rivers) have low concentrations of dissolved oxygen (DO) during periods of total ice cover, conditions which occur naturally without domestic or industrial pollution. A similar oxygen deficit was noted in some unpolluted rivers in the northern and central belt of the U.S.S.R. (Drachev, 1964). Data from various subarctic rivers in Alaska indicated that DO depletion was a continuous process throughout most of the period of ice cover, with an increase in DO concentration shortly before spring breakup (Frey, 1969; Gordon, unpubl; Schallock, unpubl). The extent of depletion increased progressively toward the lower reaches of each river (Frey, unpubl; Gordon, unpubl).

Investigations in the U.S.S.R. have shown that the low DO concentration resulted from the ice cover which prevented reaeration (Drachev, 1964). Since there is essentially no open water during the period of ice cover over many Alaskan rivers, there is little chance for significant reaeration. Under natural conditions, the extent of oxygen depletion is often sufficient to reduce the DO concentration to a level far below the 7 mg/l minimum set by the Alaska water quality standards (State of Alaska, 1967). A DO concentration of 1.1 mg/l was measured in an unpolluted arctic river (Lotspeich, unpubl; Schallock, unpubl) and, 1.1 mg/l (Gordon, unpubl) and 1.0 mg/l (Rogers, 1967) in unpolluted subarctic rivers.

The aquatic biota of Alaskan rivers seem to survive the extreme fluctuations in the amount of DO which they encounter throughout the year under natural conditions. Problems arise when oxidizable domestic or industrial wastes enter these rivers. When the biochemical oxygen demand (BOD) of these wastes is added to the natural requirement for DO, the result may be detrimental to the ecosystem.

Ingraham and Stokes (1959) discussed the numerous definitions of psychrophilic bacteria and set forth what is probably the most useful definition, "Psychrophiles are bacteria that grow well at 0° C within 2 weeks". These organisms appear to be ubiquitous in nature since they have been found in soil, rivers, lakes, mud and food (Farrell and Rose, 1967; Stefaniak, 1968; Stokes and Redmond, 1966). Psychrophiles have been studied in both the Arctic and Antarctic and have been found in soil and water (Boyd, 1958; Boyd and Boyd, 1962; Flournoy, 1967; McDonald et al., 1963; Straka and Stokes, 1960). These organisms and their activity at low temperatures have been the subject of several reviews (Farrell and Rose, 1965, 1967; Ingraham and Maaløe, 1967; Ingraham and Stokes, 1959; Witter, 1961) and will not be discussed in detail here. Stokes and Redmond (1966) considered psychrophiles to be present in large enough numbers in natural habitats to be important in the cycling of matter. Buhrmann et al. (1966) stated, "Self-purification processes start at the microbial level. . ." and, "Most of the work is accomplished by heterotrophic microorganisms (bacteria, fungi, flagellates)".

Active metabolism of organic material in a river during the winter has been demonstrated in the U.S.S.R. (Drachev, 1964). Plate counts of heterotrophic bacteria indicated that Alaskan rivers have bacterial populations in the range of 10^2 - 10^3 organisms/ml which are capable of growth on a synthetic medium at low temperatures (Gordon, unpubl). There is evidence that the number of organisms capable of growth at low temperatures increases progressively toward the lower reaches of a subarctic river in Alaska (Gordon, unpubl) and in a river in the U.S.S.R. (Drachev, 1964) during the period of total ice cover.

It has been shown that psychrophilic bacteria are capable of rapid metabolic activity at low temperatures. Since Alaskan rivers have populations of these organisms, it appears that they may be responsible for a significant portion of the DO depletion observed under both natural and polluted conditions. The subject of this report is DO depletion by the indigenous bacteria in a subarctic river. The effect of added organic and inorganic nutrients and incubation temperature on the rate and extent of DO depletion was investigated. The data obtained from a subarctic river were compared to similar data from an arctic river.

MATERIALS AND METHODS

River Description And Sampling Locations

Most of the experimental results were obtained from a subarctic river in interior Alaska. Because of the high level of domestic pollution in the lower reach and the convenient location, the Chena River was chosen for detailed study. It is a nonglacial stream with many groundwater sources, and is approximately 150 miles in length (Frey, 1969). Raw domestic sewage and effluents from seven primary sewage treatment plants in the greater Fairbanks area enter the river in the last 20 miles before it joins the Tanana River. Two sampling locations were selected, one below all major sources of domestic pollution and the other above any source.

Comparative data were obtained from an arctic river in the "Arctic Slope" area of Alaska. The Sagavanirktok (Sag) River was selected because it is the major river flowing through an area of extensive oil development and is accessible for sampling. It is a nonglacial stream originating in the Brooks Range, flowing north approximately 170 miles to the Beaufort Sea and receiving little, if any, domestic pollution (Gordon, unpubl). One sampling location was selected approximately 20 river miles above the mouth of the river near the settlement of Sagwon.

Sample Collection And Handling

All sample locations had total ice cover and a water temperature of essentially 0° C throughout the study period. Samples were obtained through holes drilled in the ice. Samples from the Chena River for the dissolved oxygen (DO) depletion study were collected in sterile five gallon polypropylene carboys by drawing water from the hole in the ice. Because of the large volume required, no problem with increase of water temperature was encountered during the 2 to 3 hour period between sample collection and handling in the laboratory.

Samples from the Sag River for the DO depletion study required somewhat different handling. These samples were collected in new, clean, but not sterile, five gallon polyethylene carboys which were sealed with tight screwcaps. It was not possible to ship the water, so it was pumped into the carboys. The samples were shipped by air freight to the laboratory. The water temperature was from approximately 0° to 3.5° C during the 9 hour period between sample collection and handling in the laboratory. This temperature rise did not appear to be excessive, and was not considered significant.

Samples for chemical analysis were collected in small mouth 250 ml screwcap polyethylene bottles. These bottles were filled by submerging them in the hole drilled in the ice and returned to the laboratory without further field treatment. They were frozen as soon as possible after arrival at the laboratory and stored at -20° C until they were analyzed.

Samples for the determination of DO in the river were collected in 300 ml biochemical oxygen demand (BOD) bottles. The bottles were lowered on a rod sampler below the bottom of the ice, allowed to fill completely, and the oxygen was fixed immediately after being brought to the surface.

Handling Of Samples In The Laboratory For DO Depletion Studies

After the samples were returned to the laboratory, they were taken directly into the 10° C cold room. The DO depletion study was set up immediately, using pre-cooled glassware to minimize any adverse effect on the natural distribution of the microorganisms in the samples. A predetermined volume of river water, 24-36 liters, and the substrate being studied were placed in a 2½ or 3½ gallon, sterile, glass carboy. The carboy was placed on the apparatus shown in Figure 1-A. The water was stirred rapidly with a magnetic stirrer while the temperature was raised 1° -

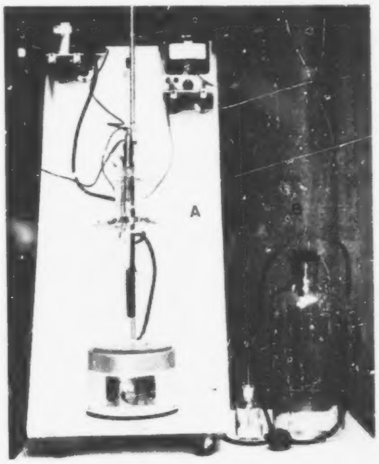


FIGURE 1 Apparatus for preparation and bottling of river water samples for dissolved oxygen depletion studies. (A) Sample was stirred vigorously while the temperature was equilibrated at 1° - 1.5° C above the desired incubation temperature with a thermostatically controlled, 1000 watt, Vicor glass, immersion heater; followed by dissolved oxygen equilibration at or near saturation by aeration with a gas dispersion tube. (B) Equilibrated sample was pumped into biochemical oxygen demand bottles for incubation.

1.5° C above the intended incubation temperature with a thermostatically controlled, 1000 watt, Vicor glass, immersion heater. The increase in temperature above that selected for incubation prevented supersaturation of the water with DO. When the desired temperature was reached, stirring was continued and the water was aerated vigorously for 10 minutes using a gas dispersion tube to bring the DO level to or near saturation. After temperature adjustment and aeration, the river water was pumped into BOD bottles as shown in Figure 1-B. The bottles were filled from the bottom to prevent entrainment of additional DO. The initial DO level was determined by immediately fixing the oxygen in three of the BOD bottles. The rest of the BOD bottles were placed in incubators at 0°, 5°, 10°, 15° or 20° C. Time intervals were selected to permit the depletion of DO to be followed. The DO was determined in three bottles at each time interval.

Substrates Used For DO Depletion Studies

Several laboratory substrates of varying complexity were used. Vitamin-Free Casamino Acids, Control 534363 (Difco) were used to compare rates of DO depletion at several temperatures, with water from various sources, and as a control for other studies. Yeast Extract, Control 523143 (Difco) and Beef Extract, Control 495576 (Difco) were used as complex substrates containing growth factors. Growth factors are defined as organic compounds, generally in minute amounts, required for growth by an organism in addition to the principal sources of carbon and energy. Glucose (Dextrose, Control 527712, Difco) was used to represent the carbohydrates. Ethyl alcohol (dehydrated, N.F., Federal Government stock no. 6505-105-0000) was the only alcohol used. Sodium acetate $NaC_2H_3O_2{\cdot}3H_2O$, Mallinckrodt analytical reagent) was used to represent the organic acids.

Primary and secondary sewage treatment plant effluents were also studied. Primary effluent was obtained from the Fairbanks city plant before the effluent entered the chlorine contact chamber. Secondary effluent was obtained from a bench scale activated sludge system being operated in the Alaska Water Laboratory at 0° - 1.0° C.

Enumeration And Isolation Of Heterotrophic Bacteria

The membrane filter method and a broth culture medium prepared from components [2.5 g/l Yeast Extract (Difco), 5 g/l Tryptone (Difco), and 1 g/l Dextrose (Difco) made up in glass distilled water and adjusted to pH 7.0 at 25° C before autoclaving] were used to enumerate bacteria at 0° C. This medium was found to give higher numbers on membrane filters at 0° C than any other medium tried (Gordon, unpubl). However, this does not mean that these were the only bacteria present in the water. All membrane filter preparation was done in the 10° C cold room, using pre-cooled equipment and materials. Incubation of filters was continued until there was no further increase in numbers on consecutive counts.

Isolation of pure cultures was accomplished by picking individual colonies from the membrane filter after the number of colonies had stopped increasing. The colonies were placed in tubes of the same broth medium used for initial enumeration, and incubated at 5° C because growth was more rapid than at 0° C. After growth appeared, material from the broth cultures was streaked on Plate Count Agar (Difco) and incubated at 5° C. Individual colonies were picked and grown in broth. This procedure was repeated as a final check of culture purity. The pure cultures were maintained for further study by monthly transfer to fresh broth and incubation at 5° C.

Chemical Analyses

The Technicon Auto Analyzer was used for the following analyses: Orthophosphate phosphorus by the Technicon ammonium molybdate industrial method; ammonia nitrogen by the sodium phenolate method (FWQA, 1969); nitrite nitrogen by diazotization (FWQA, 1969); nitrate nitrogen by hydrazine reduction (FWQA, 1969).

Total nitrogen and total carbon were determined with the Perkin Elmer model 240 Elemental Analyzer.

Total phosphorus was determined by the persulfate digestion method (FWQA, 1969), except for glucose. Glucose samples were ashed (AOAC, 1965), followed by the orthophosphate phosphorus determination previously described.

Chemical oxygen demand was determined as described in the 12th edition of **Standard Methods for the Examination of Water and Wastewater** (APHA, 1965).

DO was determined by the azide modification of the iodometric method (APHA, 1965).

Statistical Treatment Of DO Depletion Data

Each set of 3 DO measurements was evaluated by the Q Test to reject questionable results. The remaining measurements were averaged to obtain the reported result. To compare rates of DO depletion, an attempt was made to establish a rate constant with one substrate at each incubation temperature. Data obtained during the period of most rapid DO depletion were treated with first and second order kinetics, and did not fit either form. The arithmetic form, DO vs time, provided the most useful treatment of the data. A straight edge was laid along the slope of the DO depletion curve, and the data points on the portion of the curve which appeared to have the most rapid rate of change were used to establish an approximate rate (mg/l/hr) for the purpose of comparing data within this study.

RESULTS

Pure Culture Study Of Psychrophilic Bacteria Isolated From A Subarctic River

Water samples were collected from a polluted and an unpolluted location in a subarctic (Chena) river on December 17, 1968 and contained, respectively, 9000 and 550 heterotrophic bacteria per ml which were capable of growth at 0° C on the complex organic medium as described in the Materials and Methods section. All colonies on a representative membrane filter from each location were isolated in pure culture. Broth tubes inoculated with the pure cultures were incubated as shown in Table 1. All cultures from both locations grew at 0°, 5°, and 10° C, but not at higher temperatures. The percentage of the total number which did grow at 20° C and 25° C was the same from both locations. At 30° C and above, the percentage of cultures from the polluted location that grew decreased much more slowly than those from the unpolluted location. This suggested that domestic pollution caused a change in population composition.

Farrell and Rose (1967) pointed out in their review that Gram negative rods are the most common psychrophilic bacteria isolated, both qualitatively and quantitatively. Gram negative rods have been isolated from littoral and marine sediments in the Canadian Arctic (McDonald, et al., 1963) and were the most common bacteria isolated from water in subarctic Alaska (Fournelle, 1967). Farrell and Rose (1967) referred to phychrophilic members of the genus *Vibrio* (spiral bacteria) as not being as common as the Gram negative rods, but still isolated regularly.

Further study of the pure cultures revealed that only Gram negative rod and spiral morphological types of bacteria had produced colonies on the original membrane filters. The effects of incubation temperatures on the two types of bacteria from each location are shown in Table 2. The data, from the unpolluted location, indicated that increasing the incubation temperature above 25° C caused a more rapid decrease in the percentage of the spiral than of the rod shaped bacteria which grew. The results from the polluted location were similar except that the more rapid decrease of spiral bacteria took place above 30° C rather than 25° C. An additional point of interest was that all the spiral bacteria grew at 20° C, but some of the rods from both locations were inhibited at this temperature.

Examination of Table 2 shows that the ratio of rod to spiral bacteria changed from 1.5:1 at the unpolluted location to 2.9:1 at the polluted location. This twofold increase of rods relative to spiral bacteria was further indication that domestic pollution altered the composition of the bacterial population.

TABLE 1

Effect of Increased Incubation Temperature on the Growth of Bacterial Isolates from Samples Obtained from a Subarctic River[a,b]

Incubation Temperature	Sample Location			
	Polluted[c]		*Unpolluted*[d]	
	Number of Isolates	% of Total Isolates	Number of Isolates	% of Total Isolates
0° - 10° C	66	100	38	100
20	63	95.5	35	92.1
25	52	78.8	30	78.9
30	49	74.2	11	28.9
35	22	33.3	6	15.8
45	7	10.6	1	2.6
Total	66		38	

a. Samples were taken on December 17, 1968, when the river had total ice cover and the water temperature was 0° C.

b. The isolates were obtained by picking all colonies from a membrane filter which had been incubated at 0° C until there was no further increase in numbers on consecutive counts.

c. The polluted location was below a reach of the river receiving raw domestic sewage and effluents from primary sewage treatment plants.

d. The unpolluted location was upstream from any source of domestic or industrial pollution.

TABLE 2

Relative Distribution and the Effects of Increased Incubation Temperature on the Growth of Two Morphological Types of Bacteria Isolated from a Subarctic River[a,b]

Incubation Temperature	Morphological Type							
	Rod				Spiral			
	Sample Location				Sample Location			
	Polluted[c]		*Unpolluted*[d]		*Polluted*		*Unpolluted*	
	No. of Isolates	% of Isolates	No. of Isolates	% of Isolates	No. of Isolates	% of Isolates	No. of Isolates	% of Isolates
0-10° C	49	100	22	100	17	100	17	100
20	46	93.9	20	90.9	17	100	15	100
25	46	93.9	18	81.8	16	94.1	12	80.0
30	35	71.4	10	45.5	14	82.4	1	6.7
35	19	38.8	5	22.7	3	17.6	1	6.7
45	7	14.3	1	4.5	0	0.0	0	0.0
Total	49		22		17		15	

a. Samples were taken on December 17, 1968, when the river had total ice cover and the water temperature was 0° C.

b. The isolates were obtained by picking all colonies from a membrane filter which had been incubated at 0° C until there was no further increase in numbers on consecutive counts.

c. The polluted location was below a reach of the river receiving raw domestic sewage and effluents from primary sewage treatment plants.

d. The unpolluted location was upstream from any source of domestic or industrial pollution.

Effect Of Complex Organic Substrate Concentration And Incubation Temperature On The Dissolved Oxygen (DO) Depletion In Subarctic River Water

Vitamin-free casamino acids had been used previously in pure culture studies of *Pseudomonas fluorescens* at low incubation temperatures (Jezeski and Olsen, 1961; Olsen and Jezeski, 1963) and were found to give excellent growth, which was not enhanced by the addition of yeast extract. In view of these earlier reports, some preliminary results from this laboratory and the relatively simple composition of the substrate, vitamin-free casamino acids were selected as the baseline and comparative substrate.

The effect of substrate concentration on DO depletion in polluted river water is shown in Figure 2. These data indicated that a vitamin-free casamino acids concentration of 120 mg/l was sufficient to eliminate the substrate as a rate limiting factor in DO depletion. There was no lag phase at 20° C and the DO concentration in the water was reduced to nearly 0 mg/l in 15-16 hours. A similar effect of substrate concentration was observed at 10° C, but the time required to deplete the DO from near saturation to 0 mg/l was approximately 50 hours.

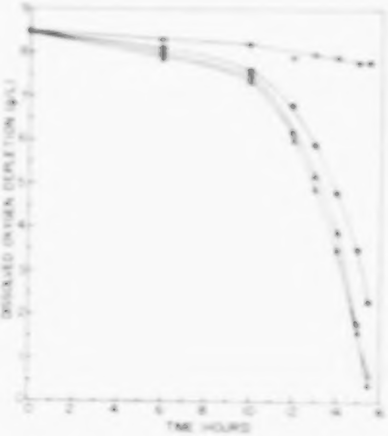


FIGURE 2 Effect of the concentration of a complex organic substrate on dissolved oxygen depletion in sub-Arctic river water polluted with raw domestic sewage and effluents from primary treatment plants. Samples were incubated at 20° C. Symbols: ●, 90 mg/l; ▲, 120 mg/l; and △, 150 mg/l vitamin-free casamino acids; o, river water blank.

Since the water temperature in the Chena River rarely, if ever, rises above 20° C (Frey, 1969; Gordon, unpubl), temperatures between 0° and 20° C were selected for incubating samples. The effect of incubation temperature on DO depletion in polluted Chena River water is shown in Figure 3. The volume of water obtained from the river was large enough to supply samples for all incubation temperatures. This provided directly comparable temperature effect data when the samples were incubated in the presence of 120 mg/l vitamin-free casamino acids. The results indicated that the length of the acceleration phase increased and the rate of DO depletion was reduced as the incubation temperature was decreased, and there was a short lag phase at the 0° C incubation temperature. However, the extent of DO depletion did not appear to be temperature dependent.

Comparative results on the effect of incubation temperature were obtained with unpolluted river water (Fig. 4). The results showed that there was a lag phase at the lower incubation temperatures (0°, 5°, and 10° C) before the acceleration phase began. This was in contrast to the lack of a lag phase with samples from the polluted location. The extent of the DO depletion, as found with the sample from the polluted location, did not appear to be temperature dependent. However, the total elapsed time was increased 50-100 percent.

The Relative Effect Of Complex Organic Substrates On DO Depletion In Subarctic River Water

Most of the psychrophilic bacteria are found in a few genera (Farrell and Rose, 1967) and some have been isolated from water in subarctic Alaska (Fournelle, 1967). Nutritional studies have shown that the growth requirements vary over a wide range, from the simple need for a carbon and energy source to the need for vitamins and other preformed growth factors (Adams and Stokes,

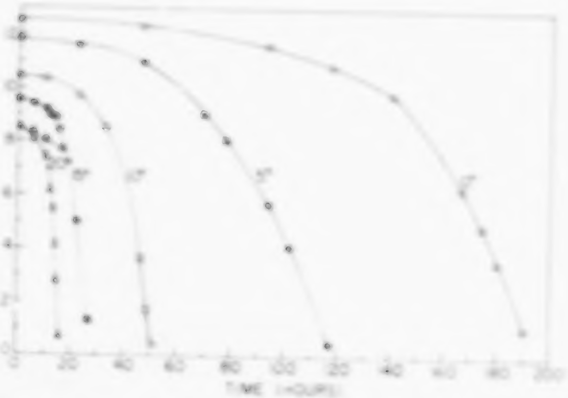


FIGURE 3 Effect of incubation temperature on dissolved oxygen depletion when 120 mg/l vitamin-free casamino acids were added to sub-Arctic river water polluted with raw domestic sewage and effluents from primary sewage treatment plants. A river water blank (▲) was incubated at 20° C.

1968; Jezeski and Olsen, 1961; Mulder, 1964; Olsen and Jezeski, 1963; Pereira and Morgan, 1957; Prince, et al., 1954). Vitamin-free casamino acids contain 18 amino acids and essentially no other growth factors, while yeast and beef extracts contain many amino acids, vitamins and other water soluble growth factors. The use of these three substrates for DO depletion studies permitted an examination of the effect of added growth factors.

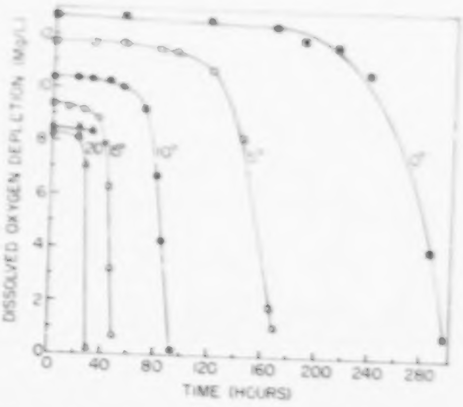


FIGURE 4 Effect of incubation temperature on dissolved oxygen depletion when 120 mg/l vitamin-free casamino acids were added to unpolluted sub-Arctic river water. A river water blank (▲) was incubated at 20° C.

All the bacteria isolated from both locations were capable of growth at 10° C on a complex medium containing a variety of preformed growth factors (Table 1), and DO depletion with river water samples took place in a reasonable time at 10° C with vitamin-free casamino acids as the substrate (Figs. 3 and 4). Since 10° C appeared to be adequate for growth and metabolic activity, it was selected as the incubation temperature for additional studies.

Water samples from both sample locations were incubated at 10° C with the three complex organic substrates in quantities containing the same amounts of carbon. The results are presented in Figures 5 and 6. The acceleration phase of the DO depletion curve was shorter with samples from both locations when the yeast or beef extract was used as the substrate. This suggested that preformed growth factors either enhanced overall metabolic activity or were required by a portion of the bacterial population. The results indicated that there was a difference in the relative effect of yeast and beef extracts on DO depletion at each location. The yeast extract caused a very pronounced decrease in the acceleration phase as related to either of the other substrates when incubated with water from the unpolluted location (Fig. 6), while the effect in water from the polluted location did not become apparent until later (Fig. 5). This could mean (a) that one or more growth factors were added with the sewage or (b) that the bacteria enhanced by domestic pollution (Table 1) did not require the growth factors in yeast extract and that those which required growth factors needed a much longer time to utilize significant DO.

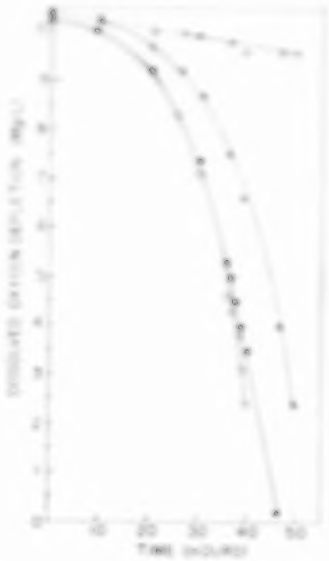


FIGURE 5 Relative effect of three complex organic substrates on dissolved oxygen depletion in sub-Arctic river water polluted with raw domestic sewage and effluents from primary treatment plants. Samples were incubated at 10° C. Symbols: ○, river water blank; △, 120 mg/l vitamin-free casamino acids; ●, 106 mg/l beef extract; □, 80 mg/l yeast extract. All three substrates contained equal amounts of carbon.

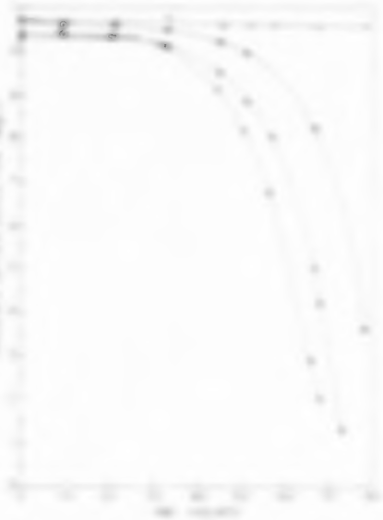


FIGURE 6 Relative effect of high levels (substrate not rate limiting) of three complex organic substrates on dissolved oxygen depletion in unpolluted sub-Arctic river water when incubated at 10° C. Symbols: o, river water blank; ●, 120 mg/l vitamin-free casamino acids; ▲, 106 mg/l beef extract; △, 88 mg/l yeast extract. All three substrates contained equal amounts of carbon.

Results similar to those obtained with a high level of substrates in unpolluted water (Fig. 6) were obtained with a low substrate level, shown in Figure 7. In all cases, this low level of substrate limited the amount of DO utilized. Growth factors added in the yeast and beef extracts shortened the acceleration phase, but the extent of DO utilization with these substrates was less than with the vitamin-free casamino acids. This suggested that one or more amino acids were required by a large portion of the bacterial population and that there was a limiting amount present in the extracts. Similar results were obtained at 0°, 5°, 15° and 20° C with this low substrate level. Since these data would be redundant, they have not been shown.

The Effect On DO Depletion When Nitrogen And Phosphorus Were Added To Subarctic River Water In The Presence Of Substrates Devoid Of These Nutrients

Ammonia, nitrite and nitrate nitrogen and orthophosphate phosphorus concentrations were determined by chemical analysis each time samples were taken from either location, and the ranges of values obtained are shown in Table 3. Both ammonia nitrogen and orthophosphate phosphorus were increased by domestic pollution.

The results in Figures 8 and 9 indicated that glucose, which contained the same amount of carbon as the vitamin-free casamino acids control, was poorly utilized as a substrate for DO depletion in Chena River water. Nitrogen and phosphorus, in amounts equal to the amounts in vitamin-free casamino acids, were added to the river water, which contained glucose. When nitrogen alone was added to the system, little effect on the DO depletion was observed with either polluted (Fig. 8) or unpolluted (Fig. 9) water. The same was true for phosphorus in the polluted water. However, when phosphorus was added to the unpolluted water, DO depletion appeared to be enhanced to some extent. This suggested that the amount of phosphorus naturally present was a limiting factor. When phosphorus and nitrogen were both added, a very marked effect on DO depletion in the

FIGURE 7 Relative effect of low levels (substrate being rate limiting) of three complex organic substrates on dissolved oxygen depletion in unpolluted sub-Arctic river water when incubated at 10° C. Symbols: ×, river water blank; ●, 20 mg/l vitamin-free casamino acids; ○, 26 mg/l beef extract; ■, 20 mg/l yeast extract. All three substrates contained equal amounts of carbon.

TABLE 3

Chemical Analysis of Water Samples from Two Locations on a Subarctic River

Determination	Range of Values (mg/liter)	
	Polluted Sample Location[a]	*Unpolluted Sample* Location[b]
Dissolved Oxygen	2.3 - 5.9	2.7 - 6.0
Ammonia Nitrogen	0.03 - 0.63	0.06 - 0.16
Nitrate Nitrogen	0.02 - 0.20	0.01 - 0.12
Nitrite Nitrogen	0.002 - 0.007	0.001 - 0.004
Orthophosphate Phosphorus	<.02 - 0.09	<0.01 - 0.02

a. The polluted location was below a reach of the river receiving raw domestic sewage and effluents from primary sewage treatment plants. The range of values is from 10 samples taken between December 16, 1969 and April 7, 1970.

b. The unpolluted location was upstream from any source of domestic or industrial pollution. The range of values is from 7 samples taken between December 16, 1969 and April 14, 1970.

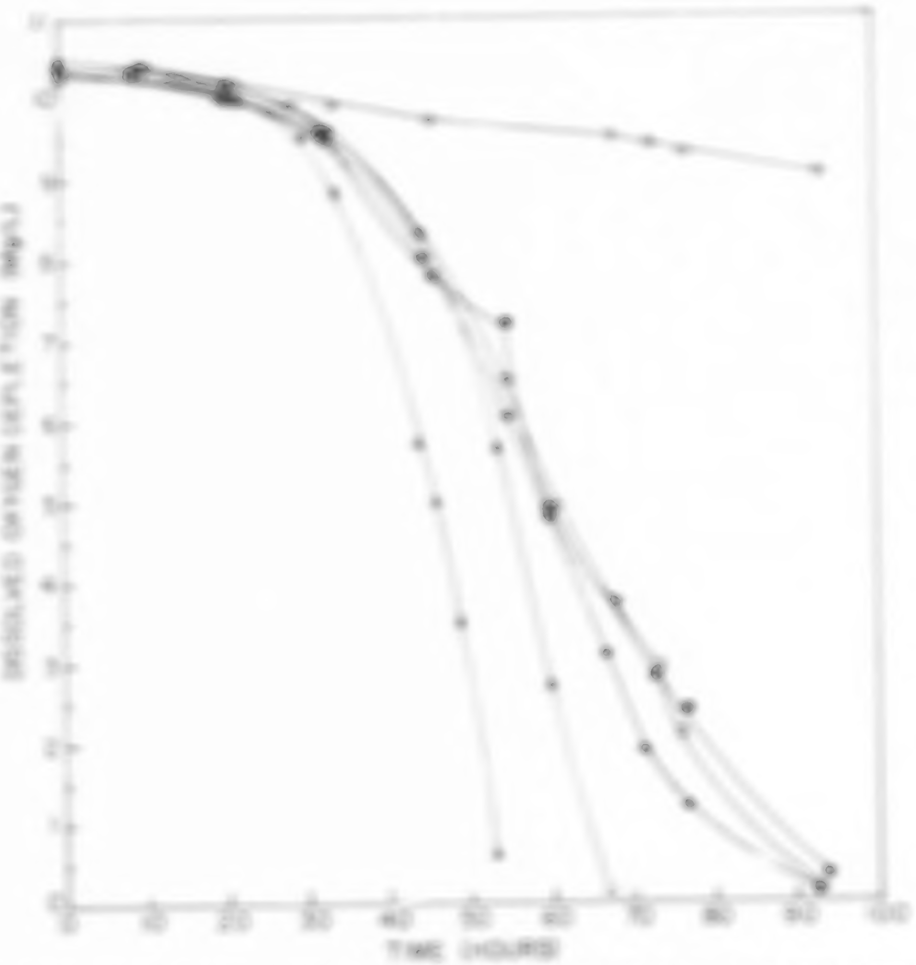


FIGURE 6 Effect of glucose on dissolved oxygen depletion in sub-Arctic river water polluted with raw domestic sewage and effluents from primary treatment plants. Incubation at 10° C in the presence and absence of added inorganic nitrogen and phosphorus. Symbols: ×, river water blank; □, 100 mg/l vitamin-free casamino acids as a control; ●, 88 mg/l glucose; ○, 88 mg/l glucose, KH_2PO_4 (0.33 mg/l phosphorus) and K_2HPO_4 (0.33 mg/l phosphorus); ▪, 88 mg/l glucose, $(NH_4)_2SO_4$ (3.33 mg/l nitrogen) and KNO_3 (10 mg/l nitrogen); ▲, 88 mg/l glucose, K_2HPO_4, KH_2PO_4, $(NH_4)_2SO_4$, and KNO_3 (nitrogen and phosphorus in same amounts as above). The glucose, K_2HPO_4, KH_2PO_4, $(NH_4)_2SO_4$, and KNO_3 were added to give the same levels of carbon, phosphorus, and nitrogen as found in the casamino acids control.

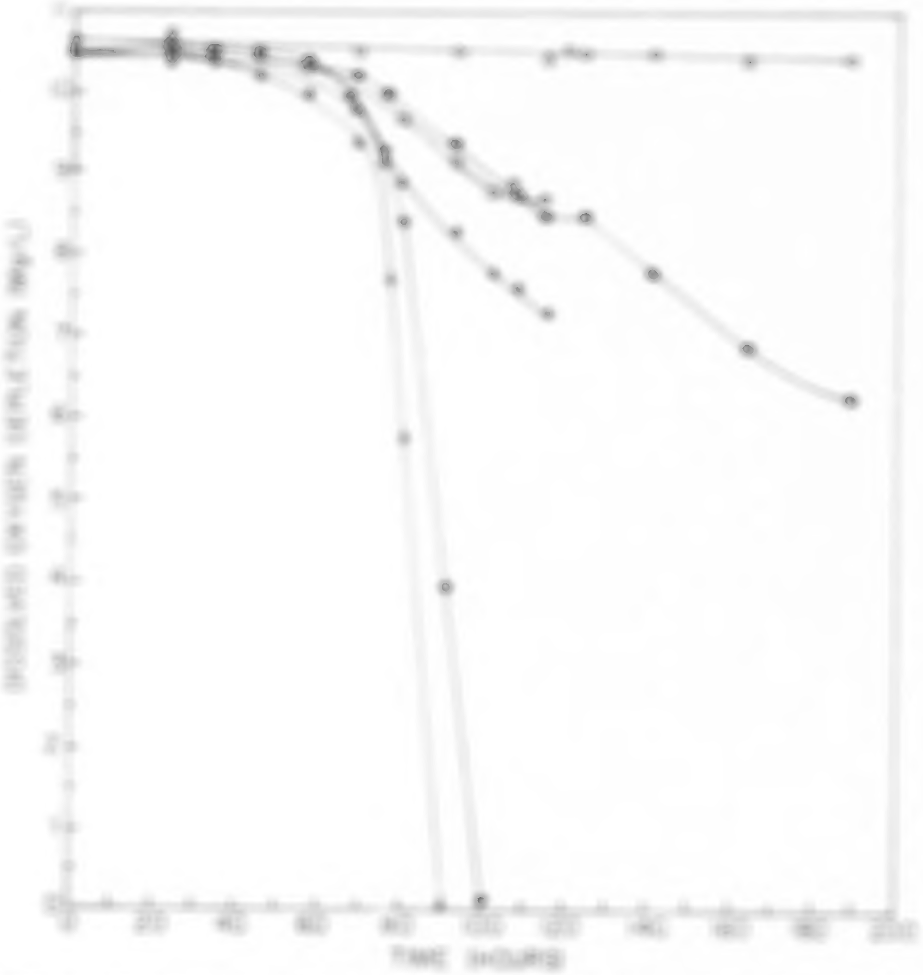


FIGURE 8 Effect of glucose on dissolved oxygen depletion in unpolluted sub-Arctic river water when incubated at [illegible] C in the presence and absence of added inorganic nitrogen and phosphorus. Symbols: x, river water blank; □, 100 mg/l vitamin-free casamino acids as a control; ●, 88 mg/l glucose; ▲, 88 mg/l glucose, KH_2PO_4 ([illegible] mg/l phosphorus) and K_2HPO_4 ([illegible] mg/l phosphorus); ○, 88 mg/l glucose, $(NH_4)_2SO_4$ ([illegible] mg/l nitrogen) and KNO_3 ([illegible] mg/l nitrogen); ◆, 88 mg/l glucose, K_2HPO_4, KH_2PO_4, and KNO_3 (nitrogen and phosphorus in same amounts as above). The glucose, K_2HPO_4, KH_2PO_4, $(NH_4)_2SO_4$, and KNO_3 were added to give the same levels of carbon, phosphorus, and nitrogen as found in the casamino acids control.

presence of glucose was observed with either polluted or unpolluted river water. This effect was more pronounced with the unpolluted (Fig. 8) than with the polluted water (Fig. 9), because it altered both the extent of DO depletion and the time span, while only the time span was changed in the polluted water. These results also suggested that a portion of the bacterial population was not active in DO depletion with glucose as the substrate, possibly because the necessary growth factors were not provided.

The effect of added nitrogen and phosphorus on DO depletion, with ethyl alcohol (Fig. 10) and sodium acetate (Fig. 11) as the substrates, was studied in unpolluted water. The carbon content of both substrates and the amount of nitrogen and phosphorus added were the same as in the vitamin-free casamino acids control. Both of these substrates were even more poorly utilized for DO depletion than was the glucose (Fig. 8) without the addition of nitrogen and phosphorus. Again, as with glucose, the addition of nitrogen and phosphorus enhanced the utilization of ethyl alcohol and sodium acetate. The DO depletion with the vitamin-free casamino acids control was still greater even though the utilization in the presence of these substrates was enhanced. This is added evidence for the role of growth factors in the metabolic activity of the bacterial population.

Both yeast and beef extracts contained slightly less ammonia nitrogen than did the vitamin-free casamino acids. The addition of ammonia nitrogen had no effect on the utilization of DO with either extract in polluted or unpolluted water, since the results were identical to those shown in Figures 3 and 4.

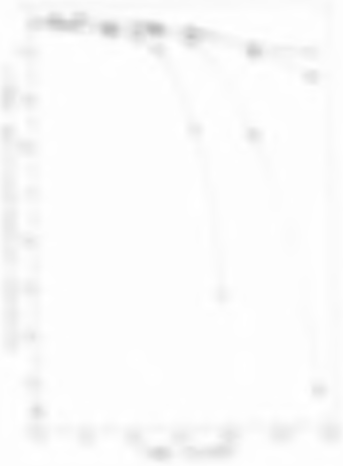


FIGURE 10. Effect of ethyl alcohol on dissolved oxygen depletion in unpolluted [illegible] river water when incubated at 20° C in the presence and absence of added inorganic nitrogen and phosphorus. Symbols: ○, river water blank; ■, 100 mg/l vitamin-free casamino acids as a control; ▲, 68 mg/l ethyl alcohol; △, 68 mg/l ethyl alcohol, K_2HPO_4 (8.53 mg/l phosphorus), KH_2PO_4 (8.53 mg/l phosphorus), $(NH_4)_2SO_4$ (12.23 mg/l nitrogen) and KNO_3 (14 mg/l nitrogen). The ethyl alcohol, K_2HPO_4, KH_2PO_4, $(NH_4)_2SO_4$, and KNO_3 were added to give the same levels of carbon, phosphorus and nitrogen as found in the casamino acids control.

[illegible]

[illegible]

[illegible]

[illegible]

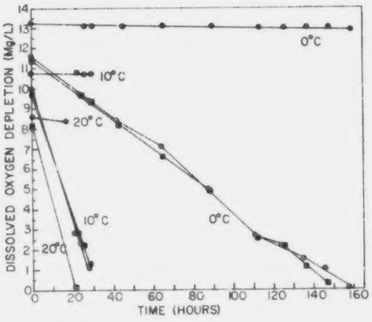


FIGURE 12 Effect of incubation temperature on dissolved oxygen depletion when effluent from the Fairbanks, Alaska city primary sewage treatment plant was added to unpolluted sub-Arctic river water. Symbols: o, 25% effluent and 75% river water; ■, 25% effluent and 75% sterile glass distilled water; ●, 25% sterile glass distilled water and 75% river water.

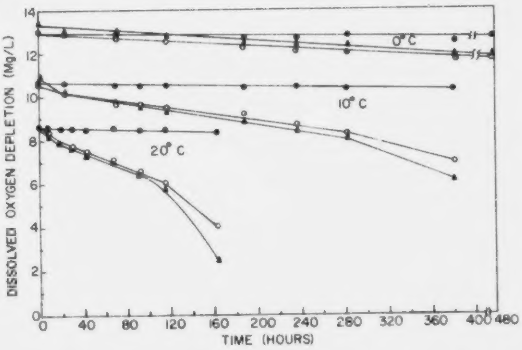


FIGURE 13 Effect of incubation temperature on dissolved oxygen depletion when effluent from a 0° - 0.5° C bench scale activated sludge sewage treatment system was added to unpolluted sub-Arctic river water. Symbols: ▲, 25% effluent and 75% river water; o, 25% effluent and 75% sterile glass distilled water; ●, 25% sterile glass distilled water and 75% river water.

Effect Of Incubation Temperature On DO Depletion In Arctic Water In The Presence Of A Complex Organic Substrate

Vitamin-free casamino acids at a concentration of 120 mg/l were used as the substrate for DO depletion studies in arctic river water (Sag River). The results, given in Figure 14, showed a lag phase at all incubation temperatures before DO depletion began. The lag phase was extremely long at the lower temperatures, particularly at 0° C. However, the extent of DO depletion did not appear to be temperature dependent. It has been shown previously (Gordon, unpubl) that the Sag

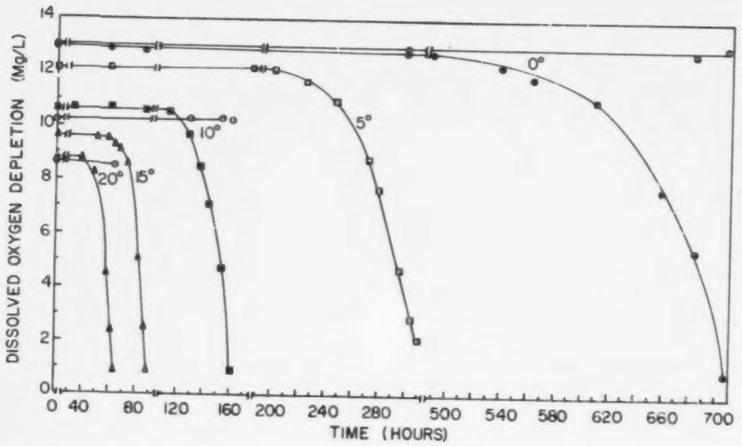


FIGURE 14 **Effect of incubation temperature on dissolved oxygen depletion when 120 mg/l vitamin-free casamino acids were added to unpolluted Arctic river water. A river water blank (o) was incubated at 0°, 10° and 20° C.**

River had a large population of heterotrophic bacteria capable of growth at low temperatures. Since only one large volume sample was available from the Sag River, the reason for the extended lag phase remains to be determined.

The same substrate concentration and incubation temperatures made it possible to relate the results from both the Sag (Fig. 14) and Chena (Figs. 3 and 4) rivers. One outstanding point was the relative time before the start of DO depletion. There was a lag phase only at the 0° C incubation temperature with samples from the polluted location on the Chena River, and the lag phase was apparent only at 0°, 5° and 10° C with samples from the unpolluted location. An extended lag phase at all temperatures was observed with samples from the Sag River. A point of similarity with all samples was that the extent of the DO depletion did not appear to be temperature dependent.

The results shown in Figures 3, 4 and 14 did not fit either the first or second order kinetic forms, so a rate constant was not obtained. Approximate rates (mg/l/hr) of DO depletion were obtained directly from the depletion curves, and the results are presented in Table 4. It must be stressed that these results are approximations and have value only in the context of this study. It would seem reasonable to have found the highest rates of DO depletion with polluted Chena River water. However, unpolluted Chena River water apparently gave higher rates than the polluted equivalent at 15° and 20° C. The rates from both Chena River samples were nearly the same at 0°, 5° and 10° C. The sample from the Sag River gave lower rates at 10°, 15° and 20° C than either Chena River sample. This suggested that the bacteria from the Sag River were more adversely affected by the higher incubation temperatures than those from the Chena River. Additional support for this suggestion was the nearly equal rates found at 15° and 20° C with Sag River water. The results showed that the source of the sample had little or no effect on the rate of DO depletion at 0° and 5° C. This suggested that all or part of the bacterial population from each source had the same ability to utilize an organic substrate at low temperatures.

TABLE 4

Comparison of the Rate of Dissolved Oxygen Depletion When a Substrate Was Added to Arctic and Subarctic River Water Samples

Incubation Temperature	Rate of Dissolved Oxygen Depletion[a] (mg/l/hr)		
	Subarctic River		Arctic River
	Polluted Sample Location[b]	*Unpolluted Sample* Location[c]	*Unpolluted Sample* Location
20°	1.36	1.73	0.62
15°	0.92	1.13	0.65
10°	0.63	0.53	0.35
5°	0.22	0.26	0.20
0°	0.22	0.25	0.20

a. 120 mg/l Vitamin-Free Casamino Acids (Difco) was added to each river water sample.

b. The polluted location was below a reach of the river receiving raw domestic sewage and effluents from primary sewage treatment plants.

c. The unpolluted location was upstream from any source of domestic or industrial pollution.

DISCUSSION AND CONCLUSIONS

A subarctic (Chena) river had a population of heterotrophic bacteria capable of growth at 0° C on a complex medium. With dissolved oxygen (DO) depletion as the measurement, there appeared to be little metabolic activity in a closed, stationary river water system. When vitamin-free casamino acids were added to the stationary system, there was rapid and extensive DO depletion at all incubation temperatures (Fig. 4). The rate of DO depletion appeared to still be increasing at the lower incubation temperatures when the oxygen was exhausted, which suggested the maximum rate had not been reached. Thus, oxygen may have been limiting.

Jezeski and Olsen (1961) found that shake cultures increased growth rate and maximum growth level of *Pseudomonas fluorescens* at 4° and 10° C as compared to stationary cultures. In the shake cultures, oxygen was no longer a limiting factor, and the bacterial cells were kept in a constantly changing micro-environment which removed metabolic end products and brought the cells in contact with new substrate. Such a dynamic system more nearly simulates environmental conditions in a river than does a stationary system. The indigenous bacteria had the potential for rapid metabolic activity in a stationary system. Next, a dynamic system must be studied to more accurately assess the role of these bacteria in the natural environment and the effects of added substrates, such as sewage effluents.

Metabolic activity observed with the protein derivative, vitamin-free casamino acids, as the substrate was not as rapid as with yeast or beef extract (Fig. 6). These extracts contained growth factors and carbohydrates in addition to proteinaceous material. It was apparent that one or more growth factors were responsible for the increased metabolic activity. However, it is not known whether the growth factors enhanced the activity of all or part of the bacterial population, or whether a portion of the population had an absolute growth factor requirement. An understanding of the role of growth factors is necessary as an aid in developing design criteria for sewage treatment plants that will provide sewage effluents that minimize the demand for DO.

In a review of psychrophilic bacteria, Ingraham and Stokes (1959) pointed out that they could carry out nearly all metabolic activities at low temperatures, but at a slower rate than at higher temperatures. Several psychrophilic and mesophilic *Arthrobacter* species were studied for effect of temperature on growth by Roth and Wheaton (1962). Rather than a sharp cut-off point, there was a continuous gradation with a decreasing lag phase at 0° C and an increasing one at 37° C. The longest lag phase they measured at 0° C was about 300 hours before the start of fairly rapid growth. They concluded that the number of generations of a specific bacterium was not temperature dependent, but the time to attain a certain number was extended at lower temperatures.

The decreasing rate of metabolic activity with decreasing incubation temperature which was reported previously (Ingram and Stokes, 1959) appeared to be borne out by the results reported here (Table 4). This was true with samples from above and below the polluted reach of the Chena River. Several significant effects on metabolic activity in the samples were noted after the Chena River had flowed through the polluted reach. The lag phase before the start of DO depletion was much shorter (Fig. 3) than with samples from above (Fig. 4), which resulted in a much shorter elapsed time from the start of incubation until all of the DO had been utilized. The apparent effect of growth factors on the rate of DO depletion was reduced (Figs. 5 and 6), and glucose was more effectively utilized as a substrate (Figs. 8 and 9). These effects on metabolic activity indicated that raw sewage and primary treatment plant effluents added a high level of organic substrates, growth factors, nitrogen and phosphorus to the river water. In addition to the nutrients, the results presented in Figure 12 showed that bacteria capable of rapid metabolic activity at low temperatures were present in the primary treatment plant effluent. Because of these factors, raw sewage and primary effluents would probably significantly increase the DO demand under ice cover.

McDonald, et al. (1963) found proteolytic bacteria in arctic littoral and marine sediments. They found that proteolytic enzymes were highly active at low temperatures and proposed that these enzymes might be significant in protein degradation in the Arctic. Rapid and extensive DO depletion was found in Chena River water at low temperatures with protein derivatives as the substrates. This suggested that proteolysis is one of the major metabolic activities of the bacteria in the Chena River.

A large variety of proteolytic bacteria have been found in sewage treatment systems (Green, 1964). Since proteolytic activity has been found at low temperatures, there are probably similar bacteria present in sewage treatment systems operating at low temperatures. Support for this suggestion was obtained from an activated sludge sewage treatment system operating at 0° C. This system reduced the DO requirements of domestic sewage to a level that appeared to be of minimal influence in Chena River water at 0° C (Fig. 13).

Earlier work with pure cultures of *Pseudomonas fluorescens* showed that glucose was a poor substrate for DO utilization at low temperatures (Jezeski and Olsen, 1961) and that the generation time was much longer than with vitamin-free casamino acids as the substrate (Olsen and Jezeski, 1963). This may have resulted from a change in the metabolic pathway of gluscose utilization at low temperatures (Jezeski and Olsen, 1961; Palumbo and Witter, 1969b), or that more glucose was consumed for cell maintenance at low temperatures (Palumbo and Witter, 1969b). Inoue, et al. (1967) found that acetate-oxidizing bacteria were vital in self-purification of rivers.

When substrates (glucose, sodium acetate and ethyl alcohol) that did not contain nitrogen, phosphorus or growth factors were added to the closed, stationary river water system, a low level of metabolic activity resulted. Adding nitrogen and phosphorus resulted in a marked increase in activity (Figs. 9, 10 and 11). Even with these nutrients present, the rate of DO depletion was greater with the vitamin-free casamino acids. This suggested that a portion of the bacterial population either required additional growth factors, or was not capable of utilizing these substrates. Even though bacteria capable of utilizing these substrates were present in the Chena River, activity would probably be at a low level because of the limited amount of nitrogen and phosphorus present under natural conditions.

Bacteria were found to have an important role in the cycling of phosphorus in the aquatic environment (Philips, 1964), and both phosphorus and nitrogen appeared to be effective in limiting metabolic activity in the Chena River under natural conditions (Fig. 9). Nitrogen and phosphorus present in raw sewage and primary sewage treatment plant effluents reduced the limiting effect of these nutrients (Fig. 8). Therefore, a method must be found to control nitrogen and phosphorus in effluents entering arctic or subarctic waters. Barth, et al. (1968) demonstrated that it is feasible to remove both nitrogen and phosphorus on a pilot plant scale using a combined chemical-biological removal system. Since several methods are available (Nesbitt, 1969), the state of the art of phosphorus removal is probably much more advanced than nitrogen removal. Perhaps the initial efforts should be directed toward adapting a phosphorus removal method.

Throughout this study, nitrogen was supplied in the form of ammonia and nitrate at the levels present in the vitamin-free casamino acids. It is necessary to determine if the nitrogen form has any effect, and what concentration is actually required. This should aid in determining what could be done to control the effect of nitrogen on receiving waters.

Results obtained with arctic river water were far too limited to be conclusive. However, there are some general similarities between the arctic (Fig. 14) and subarctic rivers (Figs. 3 and 4). More detailed study is necessary before the effects of pollutants on arctic rivers can be defined.

It is becoming increasingly obvious that the 5 day, 20° C BOD (biochemical oxygen demand) has very limited usefulness in the arctic or subarctic because receiving waters rarely reach this temperature. Previous studies by Murphy and Miller (1968), Reid and Benson (1966), and Reid (1968) showed that a 20 day BOD, incubated at a low temperature with receiving water or seed culture acclimatized at a low temperature, gave more realistic results with raw sewage. The results presented here (Figs. 12 and 13) showed that incubation temperature and diluent had an effect on DO depletion with sewage treatment plant effluents. Therefore, it is suggested that the receiving water should be used as the diluent and the incubation temperature should be at or near the temperature of the receiving water.

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Use of product and company names is for identification only and does not constitute endorsement by the U.S. Department of the Interior or the Federal Water Quality Administration.

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PREDICTION OF DISSOLVED OXYGEN LEVELS IN THE SOUTH SASKATCHEWAN RIVER IN WINTER

Robert C. Landine

INTRODUCTION

Water pollution is a problem of growing importance. The general public is becoming increasingly concerned over the desecration of our natural resources through pollution. In Canada, the Provincial and Federal Governments have acknowledged the need for preserving and upgrading our water resources and have introduced more effective legislation for pollution control.

The discharge of municipal and industrial waste water effluents into a river may lead to excessive demands on the dissolved oxygen (DO) resources of the river. To maintain the river in a satisfactory state and in proper ecological balance it is necessary to prevent the DO concentration from falling too low.

The DO concentration is regarded as one of the most important parameters for measuring or assessing the degree of pollution in a river. Water pollution control agencies always include minimum allowable DO concentrations in their list of pollution control parameters.

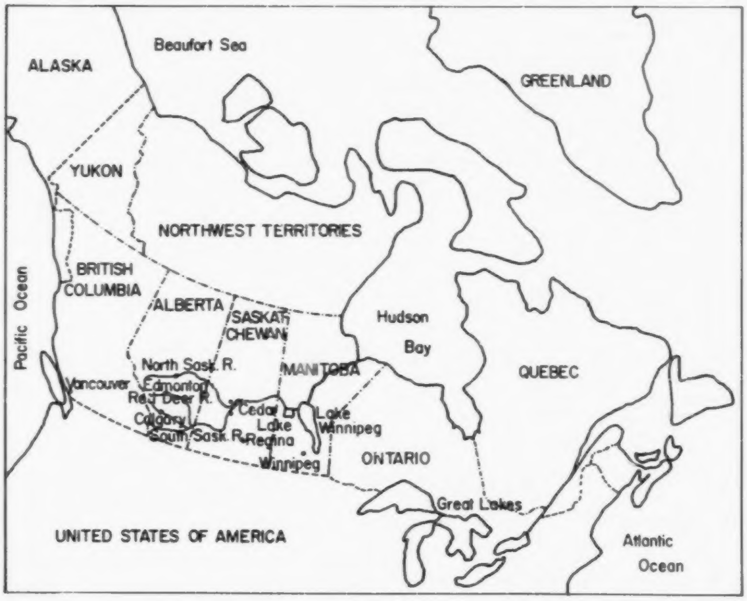
Beaufort Sea
ALASKA
GREENLAND
YUKON
NORTHWEST TERRITORIES
Pacific Ocean
BRITISH COLUMBIA
ALBERTA
SASKATCHEWAN
Hudson Bay
North Sask. R.
MANITOBA
QUEBEC
Vancouver
Edmonton
Red Deer R.
Calgary
Cedar Lake
Lake Winnipeg
South Sask. R.
Regina
Winnipeg
ONTARIO
Great Lakes
UNITED STATES OF AMERICA
Atlantic Ocean

FIGURE 1 Location of south Saskatchewan River in Canada

The DO concentration at any point along a river depends on the rates of oxygen supply and demand. The respiration oxygen demand is counterbalanced by the supply of oxygen through atmospheric reaeration and photosynthesis. The supply and demand of oxygen are related through the complex phenomenon of oxygen balance which is determined by an interplay of many physical, chemical and biological factors. Another important reason why the DO parameter is so frequently used in pollution control work is that it is possible to use mathematical equations or models to represent (approximately) the various forces operating in the oxygen balance. It is possible to formulate equations for predicting the DO level a. any point along a water course under various conditions.

The opportunity for renewal through reaeration and photosynthesis is greatly, if not entirely, reduced during the winter ice cover period which lasts five to six months in northern climates. A blanket of ice and snow over the water shuts out the light, thereby denying photosynthesis. The ice also prevents natural reaeration of the water.

From the above paragraph it follows that the oxidation of organic matter in ice covered rivers will result in a progressive reduction in the DO level as the river flows downstream. Therefore, if the oxidation rate is sufficiently high and the river is long (long time of passage) the DO concentration will eventually fall to undesirable levels. Another point is that the flows in unregulated streams in regions with prolonged frost periods tend to be minimal during winter. Therefore, the most critical DO levels may be experienced in winter rather than in summer. Almost all cases reported in the literature are for critical conditions occurring in summer.

With the distinct possibility of minimum DO levels occurring in winter coupled with the need for better methods of predicting DO levels under ice conditions, an investigation was made into the prediction of DO levels in a prairie stream under such conditions. The study was problem-oriented; it was conducted from the viewpoint that the results should be useful to a water pollution control agency.

The South Saskatchewan River was selected for a case study. It is a fairly large river, with an average annual flow of approximately 10,000 cfs, being one of two major branches which combine to form the Saskatchewan River. The Saskatchewan River System, which is 1,300 miles long, has its headwaters in the Rocky Mountains near the Alberta-British Columbia border and flows easterly through Alberta, Saskatchewan and into Lake Winnipeg, in Manitoba, as shown in Figure 1. From Lake Winnipeg, which is a large lake with an area of 9,000 mi^2, the water flows into Hudson Bay via the Nelson River. The Saskatchewan-Nelson River System is 1,600 miles long, being the second longest in the country, and forms part of the Hudson Bay drainage area basin of 1,421,350 mi^2, the largest in Canada.

The South Saskatchewan River is 900 miles long from the headwaters to the junction with the North Saskatchewan River, approximately 20 miles east of Prince Albert, Saskatchewan. From the profile shown in Figure 2 it is apparent that the river channel gradient is moderate across Saskatchewan, being approximately 1.8 ft per mile.

The South Saskatchewan River was selected for a case study because 1) it is one of the most important rivers in the region and one with which the writer had previous experience and 2) it was very convenient, passing by the University of Saskatchewan in Saskatoon. As it turned out, however, there were some aspects which rendered this river something less than an ideal choice.

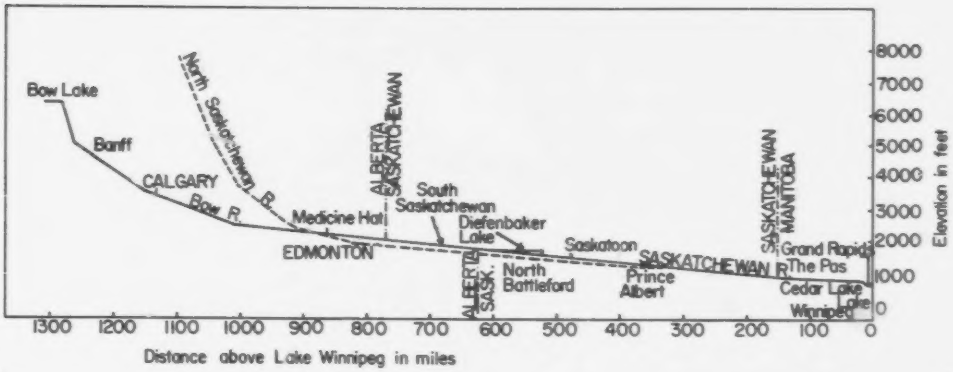

FIGURE 2 **Profile of Saskatchewan River system from headwaters to Lake Winnipeg**

Diefenbaker Lake, the large reservoir created by Gardiner Dam, formed a logical upstream boundary point, and the confluence with the North Saskatchewan River, 213 miles below the Dam, was the downstream control point. This 213-mile reach of river received pollution loadings from municipalities and industries and much greater loadings were anticipated in the future.

The 213-mile reach selected for a case study was subdivided into 11 reaches as shown in Figure 3.

DEVELOPMENT OF DO EQUATIONS (MODELS)

Ideally, a model should be developed from basic considerations, checked against field observations and modified so that it will reproduce an existing situation. Then, presumably, the model could be used to predict what would happen to the DO profile if certain changes, such as river flow and waste loading, occurred in the river system.

The great amount of human and financial resources that would have been required precluded the possibility of evaluating the various factors and coefficients from actual field surveys. Therefore, it was decided that the coefficients would have to be determined from a literature search supported by laboratory investigation and field surveys.

Time does not permit a description of the laboratory experiments conducted and the treatment of data resulting therefrom. Only the pertinent results will be referred to as required in discussing the oxygen balance situation for each reach.

Two worst winter situations were considered, viz., (1) future and (2) the conditions existing in 1968-69. Only the latter case will be considered in this paper. By worst situation is meant a prolonged period of subzero weather coupled with low barometric pressure and calm conditions.

Two different river flows were considered, 4,500 cfs and 11,000 cfs. A flow of 4,500 cfs was possible and would occur when one of the three turbines was operating at the hydroelectric station at Gardiner Dam; this would be a minimum winter flow. A flow of 11,000 cfs occurs when all three turbines are in operation. Also, a flow of 11,000 cfs was being released from Gardiner Dam at the time the writer made a field survey to check the DO level, and other parameters, downstream from the Dam. It was therefore useful to include a flow of 11,000 cfs (although such a high flow would not produce a worst condition) so that the predicted DO profile could be checked against the observed profile. There were no tributaries and, thus, the same flow applied over the entire reach.

The input waste loadings and station mileages are given in Table 1.

A discussion of the important factors considered in developing the equations to calculate the DO profile is given below.

INITIAL CONDITIONS

The initial DO concentration of the water entering the river at the exit from the Dam was taken as 11.5 ppm, the value observed on two occasions.

The initial value of the equilibrium saturation value for oxygen dissolved in water at 0° C was taken as 14.63 ppm based on the data presented by Montgomery, et al. (1964). Only one temperature was used as it was assumed that the water temperature in both the open water reaches and the ice covered reaches could be taken as 0° C. Only a minimal error was inherent in this assumption. A station pressure correction was required. Based on observations made by the

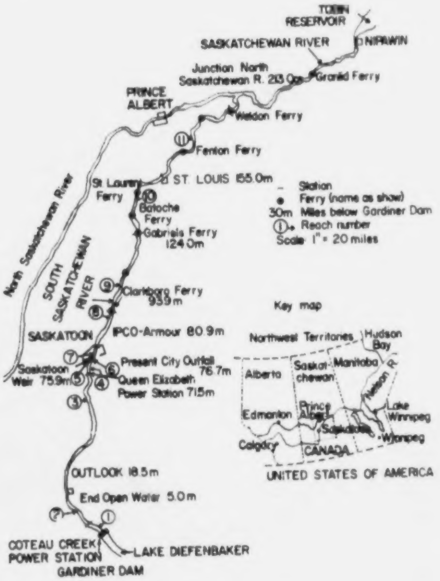
TOBIN RESERVOIR
SASKATCHEWAN RIVER
NIPAWIN
Junction North Saskatchewan R. 213.0m
Grand Ferry
PRINCE ALBERT
Weldon Ferry
⑪
Fenton Ferry
St Laurent Ferry
ST. LOUIS 155.0m
⑩
Batoche Ferry
Gabriels Ferry 124.0m
North Saskatchewan River
SOUTH SASKATCHEWAN RIVER
⑨
Clarkboro Ferry 93.9m
⑧
SASKATOON
IPCO-Armour 80.9m
⑦
Present City Outfall 76.7m
⑥
Saskatoon Weir 75.9m
⑤
④
Queen Elizabeth Power Station 71.5m
③
OUTLOOK 18.5m
End Open Water 5.0m
②
①
COTEAU CREEK POWER STATION
LAKE DIEFENBAKER
GARDINER DAM
Station
Ferry (name as show)
30m Miles below Gardiner Dam
① Reach number
Scale 1" = 20 miles
Key map
Northwest Territories
Hudson Bay
Alberta
Saskatchewan
Manitoba
Nelson R.
Edmonton
Prince Albert
Saskatoon
Lake Winnipeg
Winnipeg
Calgary
CANADA
UNITED STATES OF AMERICA

FIGURE 3 South Saskatchewan River from Gardiner Dam to confluence with North Saskatchewan River

Saskatchewan Research Council (1969) a typical winter low pressure would be 698 mm of mercury. No allowance was made for the differences in station pressure due to the difference in elevation of the open water reaches at the Dam and 70 miles downstream at Saskatoon. The difference in station pressures at the two locations was so small that it was satisfactory to consider only one saturation value, i.e., 14.63 x 698/760 = 13.45 ppm for both locations.

TABLE 1

Station Description and Waste Loadings

Station Name	Station Number	Reach Number	Station Mileage	BOD_5 Input in Pounds Per Day
Gardiner Dam	1	1	0.0	0
End Open Water	2	2	5.0	0
Outlook	3	3	18.5	280
Q.E. Power Station	4	4	71.5	0
Weir Upstream	5	5	75.9	0
Weir Downstream	6	6	75.9	0
Present City Outfall	7	7	76.7	30,000
IPCO-Armour	8	8	80.9	300
Clarkboro Ferry	9	9	93.9	0
Gabriel's Ferry	10	10	124.0	0
St. Louis	11	11	155.0	40
Weldon Ferry	12		202.0	0

Reach 1, Gardiner Dam To End Open Water, Mile 0.0 To 5.0

The river remains open, even during the coldest weather, below Gardiner Dam as a result of the water being discharged from the hydroelectric station at the Dam.

The lowest rates of reaeration would occur during low pressure, calm conditions. It was assumed that in the worst situation both of these conditions could occur in the open water reach 1 and again when the corresponding volume of water arrived at the open water reaches in Saskatoon.

The minimum area of open water exposed to the atmosphere would occur at the end of a prolonged period of subzero weather. Under these conditions it was estimated that the river would be open over the entire channel width for a distance of 5 miles below the Dam.

The following equation developed by Owens, et al. (1964)

$$k_2 = 9.41V^{0.67}h^{-1.85} \qquad (1)$$

where k_2 = reaeration coefficient, day^{-1}, base 10

V = river velocity, fps

h = river mean depth, ft

was used to calculate the reaeration coefficient. No reduction was made for the effect of dissolved salts, detergents or other contaminants since the concentration of all of these would be low, resulting in a negligible correction to k_2.

As is evident from Equation 1, k_2 is a function of velocity and depth. Since the input data was in terms of flow, Q, rather than h or V, it was necessary to express the latter two variables as a function of Q. By plotting the data presented by Tywoniuk (1969) the following equations of best fit were derived for reach 1

$$V = 0.11Q^{0.32} \tag{2}$$

and

$$h = 0.072Q^{0.47} \tag{3}$$

Likewise, the data given by Schriek (1963) was used to derive the following equations for reach 4, which was partly ice free

$$V = 1.55Q^{0.77} \tag{4}$$

$$h = 3.10Q^{0.095} \tag{5}$$

It may be noted here that a relationship giving k_2 as a function of Q may be obtained by substituting Equations 2 and 3, or 4 and 5, into Equation 1. This will result in the following equations for reach 1 and reach 4

$$k_2 = 286Q^{-0.67} \text{ (reach 1)} \tag{6}$$

$$k_2 = 0.015Q^{0.34} \text{ (reach 4)} \tag{7}$$

From Equation 6 it is apparent that for reach 1 k_2 is inversely proportional to $Q^{2/3}$, i.e., k_2 increases as Q decreases which is very helpful from the water quality point of view. However, the opposite situation is true for reach 4 which is atypical due to a weir located at the downstream end of the reach. In this reach the reaeration rate increases as the flow increases.

Past experience (Lackie and Sparling, 1955; Sparling, 1957; Bouthillier, 1968; SWRC, unpubl; Landine, 1962) with prairie rivers under ice cover conditions has shown that there is always a certain background or residual BOD_5 of the order of 1 to 2 ppm, in waters far removed from significant waste water discharges. For example, the BOD_5 of South Saskatchewan River water upstream from Saskatoon was 1.0 ± 0.8 ppm (based on 18 samples collected in the months of December to February from 1957 to 1969). The only significant sources of municipal or industrial pollution were located hundreds of miles upstream.

The explanation of a residual BOD is believed to be related mainly to the slow oxidation of humus and, to a lesser degree, to the oxidation of anaerobic end products. Humus refers to that portion of the organic matter that is very resistant to biological decomposition and could yield an essentially constant daily BOD for several days if the amount of humus were sufficient and the period of time involved not excessive. Periods of several months have been reported for complete stabilization of polluted river water even at the normal temperature of 20° C (Gannon, 1963; Gameson, 1958).

The humus material has its origin in 1) the waste water effluents discharged from industries and municipalities in Alberta (mainly) and Saskatchewan and 2) the variety of organic materials entering the river through land drainage. Land drainage occurs only during summer. However, due to the long flow-through time in Diefenbaker Lake (1 year storage capacity) the effect is felt throughout the winter period both in the lake and in the water released from it.

There would be a certain amount of anaerobic decomposition in the vicinity of the benthic growths or muds; this would be particularly true in the lake. The oxidation of anaerobic end products creates an oxygen demand.

The residual oxygen demand (ROD) was treated as a constant demand, in ppm per day, applied over the entire 213 mile reach. The magnitude of ROD was obtained from the above mentioned BOD_5 @ 20° C, (i.e. 1.0 ± 0.8 ppm) as follows

$$\text{ROD ppm/day} = \frac{\text{BOD}_5 \text{ @ } 20^\circ\text{ C}}{5 \text{ days} \times \text{FACT}} \qquad (8)$$

where FACT is the factor for converting from BOD_5 @ 20° C to the BOD_5 @ 0° C for polluted river water. From experiments conducted by Landine (1970) it was found that the magnitude of FACT was between 3.0 and 3.5 for polluted river water samples. From Equation 8 the mean value of ROD was 0.062 ppm per day; the upper and lower values were 0.120 and 0.011 ppm per day respectively.

There were no municipal or industrial discharges in reach 1.

There was no need to consider organic load contributions due to local run-off along the entire 213 mile reach because there is no run-off during severe winter conditions.

Photosynthesis was assumed to be insignificant in the oxygen economy of the river in the open water reaches as well as the ice covered reaches. This assumption was believed to be reasonable in view of the low temperature, the few hours of daylight, and the low intensity of solar radiation in winter. Table 2 illustrates that the solar radiation is much lower in winter than in summer in the area.

TABLE 2

Solar Radiation at Saskatoon, 1955 to 1967
(Saskatchewan Research Council)

Jan.	Feb.	Mar.	Apr.	May	Jun.	Jul.	Aug.	Sept.	Oct.	Nov.	Dec.
102*	199	326	414	508	544	568	466	326	197	103	74

The average annual total = 115 kilolanglies

* Units are langlies/day, equivalent to g cal/cm^2 day.

After evaluating the various factors of importance in reach 1, the DO equation was written. The deficit at the end of reach 1 is a function of two forces, viz., 1) reaeration and 2) the constant ROD rate. The differential equation may be written as

$$\frac{dD_t}{dt} = -K_2 D_t + ROD \qquad (9)$$

and integrated to yield

$$D_t = D_o e^{-K_2 t} + ROD\,(1 - e^{-K_2 t})/K_2 \qquad (10)$$

where

D_t = DO deficit at time t, i.e., at end of reach,

D_o = DO deficit initially, i.e., at beginning of reach,

t = time of flow in reach in days, obtained from Equation 2 and 4. (For ice covered reaches the flow time was increased by 30% to allow for the reduced velocity of flow.)

K_2 = reaeration coefficient, day^{-1}, base e, and

ROD = constant residual oxygen demand rate @ 0° C, ppm/day.

It was unnecessary to write a partial differential equation because with no photosynthesis there would be no diurnal fluctuation in the DO concentration; also, the waste loadings were treated as constant throughout the day. A term for dispersion was also neglected; dispersion has a negligible effect on the calculated DO profile in freshwater streams (Dobbins, 1964).

Reach 2, End Open Water To Outlook, Mile 5.0 to 18.5

There was no reaeration and no waste loading in this reach. The change in DO concentration was a function only of ROD. Equation 11 was used to calculate the deficit at the end of reach 2

$$D_t = D_o + ROD\,(t) \qquad (11)$$

where each term is as defined previously; a complete list of symbols may be found in Appendix 1.

Reach 3, Outlook to Queen Elizabeth Power Station, Mile 18.5 to 71.5

Since organic matter will remain in suspension if the stream velocity is more than 1.0 fps (Imhoff, et al., 1956) it was assumed that sludge deposits need not be considered in this river in which the velocity exceeds 1 fps. Furthermore, most of the solids which would settle out are removed in primary sedimentation and no waste water may be discharged without first receiving at least primary sedimentation or its equivalent (SWRC, 1967).

A small amount of municipal waste is contributed by the town of Outlook.

The forces causing the DO deficit to increase in this reach are the ROD and the first-order BOD arising from the Outlook sewage discharge. Based on field and laboratory tests conducted by Landine (1970) there was no need to allow for a nitrification oxygen demand because nitrification did not occur at 0° C.

The oxygen demand due to the waste water discharge may be calculated from Equation 12 which is the usual first-order BOD equation.

$$y_t = L\ (1 - e^{-K_1 t}) \tag{12}$$

where y_t = oxygen demand after t days, in ppm,

L = ultimate first-stage BOD, ppm,

K_1 = first-order BOD rate constant, day^{-1}, base e.

The sequence of calculations was as follows:

(1) the waste discharge in lb BOD_5 @ 20° C per day (raw data, see Table 1) was converted to ppm BOD_5 @ 20° C in the river by dividing by 5.38Q;

(2) the river BOD_5 @ 20° C was converted to a BOD_5 @ 0° C by dividing by FACT for which values of 3.0 and 3.5 were used;

(3) the river BOD_5 @ 0° C was converted to an ultimate first-order BOD at 0° C by Equation 12;

(4) The ultimate BOD from step (3) was added to the ultimate BOD remaining at the end of the previous reach (which was zero for reach 3 but not for subsequent reaches) to get the total ultimate BOD @ 0° C at the beginning of the reach;

(5) the BOD in the reach was calculated by Equation 12,

(6) the BOD calculated in step (5) was subtracted from the total ultimate BOD at the beginning of the reach to calculate the ultimate BOD remaining at the end of the reach.

To do step (3) above it was necessary to know the value of K_1. The exact value of K_1 in the river was not known. To overcome this problem a range of K_1 values, from 0.02 to 0.6 day^{-1}, was used in calculations. The lowest value 0.02, is very low and approximates a linear or constant oxygen demand rate whereas 0.6 is believed to be as high as can be expected. Bouthillier (1968) reported that the oxidation rate constant may be as high as 0.6 in the North Saskatchewan River below Edmonton; Cameron (1967) reported the presence of slimes on the bottom in this reach which could have accounted, in part at least, for the high rate of oxidation.

The following equation was used to caluclate the DO deficit at the end of reach 3

$$D_t = D_o + ROD(t) + YREACH \tag{13}$$

where YREACH = first-order BOD exerted in the reach due to waste water contributions.

Reach 4, Queen Elizabeth Power Station to Weir, Mile 71.5 to 75.9

The cooling water from two power stations maintains a strip of open water along one side of the river in this reach. The width of the open water strip depends on such factors as the river flow, duration of severe cold spell and distance below cooling water discharge point. It was estimated that under the worst conditions assumed for this study 15% of the river channel would be ice free.

This reach was treated as two rivers flowing side by side separated by an imaginary wall to prevent mixing back and forth. Since the reach was just 4.4 miles long with 85% of the channel width covered with ice the above assumption was believed to be reasonable.

Reaeration, first-order BOD and ROD were the important factors considered in the open water river. The differential equation was

$$\frac{dD_t}{dt} = K_1 L + ROD - K_2 D_t \tag{14}$$

which may be integrated to

$$D_t = D_o e^{-K_2 t} + \frac{K_1 L_o}{K_2 - K_1} (e^{-K_1 t} - e^{-K_2 t}) + \frac{ROD}{K_2} (1 - e^{-K_2 t}) \tag{15}$$

For the ice covered strip of river the forces considered were ROD and first-order BOD, and Equation 13 was used to calculate the deficit at the end of the reach.

Reach 5, The Weir, Mile 75.9

The amount of aeration realized at this ogee-shape weir was calculated by means of Equation 16, after Gameson, et al. (1958).

$$r = 1 + 0.11ab(a + 0.046T)h_t \tag{16}$$

where r = ratio of DO deficit above and below weir,

a = coefficient dependent on nature of water,

b = coefficient dependent on nature of weir, and

h_t = fall height at weir in feet.

It was estimated that the value of a would be 1.25 because the river was 'clean' at this point. Values of the coefficient b for an ogee-shape weir were not found in the literature; two alternate values, 0.5 and 0.75, were estimated for the value of b in the computations.

The water fall height, h_t, must be known to use Equation 16. It was found that a semilog plot of h_t versus Q linearized (Fig. 4) to yield the equation

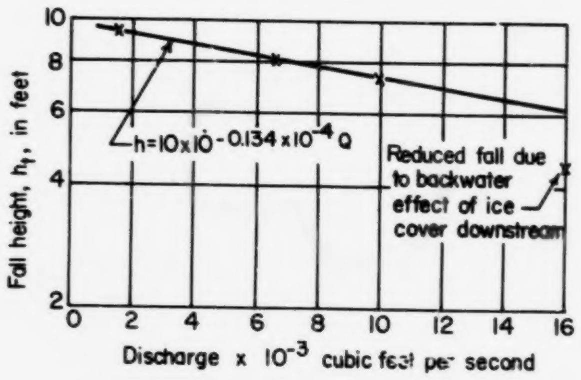

FIGURE 4 Fall height at Saskatoon Weir

$$h_t = 10 \times 10^{-0.134 \times 10^{-4} Q} \tag{17}$$

Equation 17 was based on data for open water conditions and a correction was necessary to allow for the reduced fall height due to the backwater effect of the ice cover below the weir. Limited field observations suggested a correction of minus 2 ft so that the equation used to calculate h_t as a function of Q was

$$h_t = 10 \times 10^{-0.134 \times 10^{-4} Q} - 2 \tag{18}$$

Reach 6, Weir To Present Saskatoon Outfall, Mile 75.9 To 76.7

It was estimated that the river would be 100% ice covered at mile 76.7 and that 10% of the river channel width would be open water in reach 6. Since the flow time in this reach was very small, the differential equation could be left in the differential form, without introducing serious error, to calculate D_t. The forces considered were (1) ROD, (2) first-order BOD and (3) reaeration (in the open water strip). The equation used was

$$dD_t = K_1 L\,dt - 0.1 K_2 D_o\,dt + \text{ROD}\,dt \tag{19}$$

where dD_t is the change in DO deficit in passing through the reach during a flow time equal to dt.

Reach 7,8, 9, 10 & 11, Mile 76.7 To 213

All of these reaches were treated in the same fashion because they were completely covered with ice. Waste loadings were considered when necessary. (See Table 1 for amount and location.) The two factors of significance were first-order BOD due to waste water effluents and ROD; Equation 13 was used to calculate the deficits at the end of each reach.

The prediction of the DO profile along the 213 mile reach was done with the aid of a digital computer.

DISCUSSION OF COMPUTER SOLUTIONS OF DO EQUATIONS

It was not possible to include all of the computed output as this involved several hundred lines; therefore only a few selected iterations have been included to illustrate the main points of interest as discussed below.

To check the effect of changing the variables Q, FACT, b, K_1 and ROD, a suitable range of values was used for each one. This was written into the computer programs, using a nest of DO loops, resulting in many iterative calculations for the various combinations. By scanning through the output (which involved less than 2 min. computer time) the reader was quickly able to determine which factors had the greatest significance in regard to the minimum DO concentration. This was a low cost and rapid method of evaluating numerous alternatives.

The first point of interest centered around the question of whether or not the predicted profile agreed with the observed profile.

The output showed that, when Q = 11,000 cfs, ROD = 0.011 ppm/day, FACT = 3.0 or 3.5, b = 0.5 and for any value of K_1 from 0.02 to 0.6 day^{-1}, the calculated DO concentration at station 12* was 12.0 ppm, which agreed very closely with the value of 11.9 ppm found on March 4, 1969. For the same conditions, except with ROD changes from 0.011 to 0.062 and 0.12, the calculated DO concentration at station 12 was 11.65 and 11.3 ppm respectively. The predicted values for the three different values of ROD and the observed values are shown in Figure 5.

At first glance it appeared that the predicted profile was poorer for the higher values of ROD (i.e. 0.062 and 0.12), considering the furthest station downstream as the governing or most critical point. However, a closer look showed that the predicted profile downstream from Saskatoon approximately paralleled the observed profile for the case where ROD = 0.12 ppm per day, suggesting that this upper value of ROD was the most appropriate one to use.

The discrepancies between the predicted and observed concentrations (which were 0.8 ppm or less) would have been reduced if the field survey could have been made under the same conditions as were assumed for the model. For example, due to the mild weather prevailing during the field survey (March 4 and 5, 1969) the area of open water in reaches 1, 4 and 6 was greater than for the worst conditions assumed in the model. Also, instead of the assumed calm conditions, there was a breeze ranging from 5.9 to 22.0 mph (mean daily), prevailing for the period February 22 to March 5, 1969 (SRC, 1969). The effect of the wind would have been to increase the reaeration rate, a phenomenon which has been reported by others (Anon., 1964; Bohnke, 1966). Finally, the barometric pressure was 15 to 28 mm of mercury greater over this same period than was assumed in the model. All three of the above factors resulted in higher reaeration rates than for the assumed conditions in the model and thus helped to explain why the observed concentrations were higher than the predicted DO concentrations.

Based on the above mentioned results, it was felt that the oxygen balance forces and equations employed were sufficiently representative to be useful in making predictions for this river.

*Station 12 was at Weldon, 202 miles below Gardiner Dam, the last sampling station before this river flowed into the Saskatchewan River.

A number of conclusions or deductions could be made from the computed output for winter conditions.

(1) The magnitude of the value employed for ROD had a more pronounced effect on the results than did the magnitude of K_1. For example, for the minimum flow situation of 4,500 cfs, changing ROD from 0.011 to 0.12 changed the DO concentration at station 12 from 11.93 to 10.98 ppm, i.e., 1 ppm, whereas changing K_1 from 0.02 to 0.6 changed the concentration from 11.93 to 12.02 ppm, i.e., 0.1 ppm.

(2) The magnitude of K_1 had little effect on YREACH and, consequently, on the DO profile. As illustrated in column 15 of Table 3, when the other variables were held constant, changing K_1 from 0.02 to 0.60 day^{-1} resulted in the sum of the first-order BOD changing from 0.53 to 0.43 ppm which is minor. From column 13 it can be seen that the value of the ultimate BOD at the beginning of a reach is much higher for the case of K_1 = 0.02 than for K_1 = 0.60. However, YREACH changed only slightly because a high value of ultimate BOD was used in conjunction with a low value of K_1 to calculate YREACH (using Equation 12) and vice versa, which produced a compensation effect. This suggested that it is not necessary to have an 'exact' value of K_1 for calculating the first-order BOD in the river provided that the BOD_5 @ 0° C of the waste water loadings are known.

(3) The oxygen added in open water reaches and at weirs is highly significant in the oxygen economy of ice covered rivers. This affords an opportunity for the oxygen concentration to be increased considerably in a short time. The oxygen thus added becomes available for the slow oxidation of organic matter in subsequent ice covered reaches.

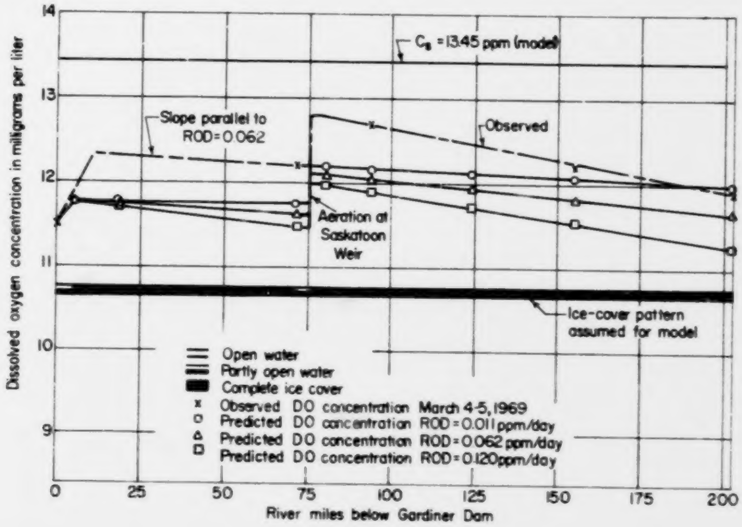

FIGURE 5 Comparison of observed and calculated DO levels for winter conditions, Q = 11,000 cfs.

TABLE 3

Partial Computed Output for 1968-69 Winter Conditions

1 STA	2 I	3 BOD	4 DK	5 ROD	6 BWEIR	7 FACT	8 C	9 D	10 V	11 H	12 F	13 ULTROB	14 ULTROE	15 YREACH	16 TIME	17 TO TIME
DK = 0.02				ROD = 0.011				FACT = 3.0				BWEIR = 0.50				C = 4500
0.0	1	0.	0.02	0.011	0.50	3.0	11.50	1.95	1.10	3.91	1.48	0.00	0.00	0.00	0.25	0.00
5.0	2	0.	0.02	0.011	0.50	3.0	12.09	1.36	1.10	3.91	1.48	0.00	0.00	0.00	0.67	0.25
18.5	3	280.	0.02	0.011	0.50	3.0	12.09	1.36	1.10	3.91	1.48	0.04	0.04	0.00	2.62	0.91
71.5	4	0.	0.02	0.011	0.50	3.0	12.06	1.39	0.68	6.89	0.38	0.04	0.04	0.00	0.35	3.53
75.9	5	0.	0.02	0.011	0.50	3.0	12.08	1.37	1.10	3.91	1.48	0.04	0.04	0.00	0.00	3.88
75.9	6	0.	0.02	0.011	0.50	3.0	12.51	0.94	1.10	3.91	1.48	0.04	0.04	0.00	0.04	3.88
76.7	7	30000.	0.02	0.011	0.50	3.0	12.52	0.93	1.10	3.91	1.48	4.38	4.36	0.02	0.21	3.92
80.9	8	300.	0.02	0.011	0.50	3.0	12.50	0.95	1.10	3.91	1.48	4.40	4.35	0.06	0.64	4.12
93.9	9	0.	0.02	0.011	0.50	3.0	12.43	1.02	1.10	3.91	1.48	4.35	4.22	0.13	1.49	4.77
124.0	10	0.	0.02	0.011	0.50	3.0	12.29	1.16	1.10	3.91	1.48	4.22	4.09	0.13	1.53	6.25
155.0	11	0.	0.02	0.011	0.50	3.0	12.15	1.30	1.10	3.91	1.48	4.09	3.91	0.19	2.32	7.78
202.0	12	0.					11.93	1.52				3.91		0.53		10.10
DK = 0.60				ROD = 0.011				FACT = 3.0				BWEIR = 0.50				Q = 4500
0.0	1	0.	0.60	0.011	0.50	3.0	11.50	1.95	1.10	3.91	1.48	0.00	0.00	0.00	0.25	0.00
5.0	2	0.	0.60	0.011	0.50	3.0	12.09	1.36	1.10	3.91	1.48	0.00	0.00	0.00	0.67	0.25
18.5	3	280.	0.60	0.011	0.50	3.0	12.09	1.36	1.10	3.91	1.48	0.00	0.00	0.00	2.62	0.91
71.5	4	0.	0.60	0.011	0.50	3.0	12.06	1.39	0.68	6.89	0.38	0.00	0.00	0.00	0.35	3.53
75.9	5	0.	0.60	0.011	0.50	3.0	12.08	1.37	1.10	3.91	1.48	0.00	0.00	0.00	0.00	3.88
75.9	6	0.	0.60	0.011	0.50	3.0	12.51	0.94	1.10	3.91	1.48	0.00	0.00	0.00	0.04	3.88
76.7	7	30000.	0.60	0.011	0.50	3.0	12.52	0.93	1.10	3.91	1.48	0.44	0.38	0.05	0.21	3.92
80.9	8	300.	0.60	0.011	0.50	3.0	12.46	0.99	1.10	3.91	1.48	0.39	0.26	0.12	0.64	4.12
93.9	9	0.	0.60	0.011	0.50	3.0	12.33	1.12	1.10	3.91	1.48	0.26	0.11	0.16	1.49	4.77
124.0	10	0.	0.60	0.011	0.50	3.0	12.16	1.29	1.10	3.91	1.48	0.11	0.04	0.07	1.53	6.25
155.0	11	0.	0.60	0.011	0.50	3.0	12.08	1.37	1.10	3.91	1.48	0.04	0.01	0.03	2.32	7.78
202.0	12	0.					12.02	1.43				0.01		0.43		10.10

NOTE: DK and BWEIR correspond to K_1 and b respectively in script.

The figures in column 8 of Table 3 show that the oxygen level increased by (12.09-11.50) = 0.59 ppm in just 5 miles of open water in reach 1, which is more than the cumulative first-order BOD in all reaches. In the same column, the results show that the amount of oxygen added through aeration at the weir was approximately equal to the cumulative first-order BOD of the wastes added in all reaches. It may be noted that more DO would have been added in reach 1 and at the weir if the deficit had been greater.

By consideration of the basic forces of oxygen supply and demand it was possible to write a series of DO equations which satisfactorily represented the oxygen balance in the South Saskatchewan River under winter conditions.

It was found that the magnitude of the first-order deoxygenation rate constant was not critical in evaluating DO levels when the BOD_5 of the waste water loadings @ 0° C was known. The results illustrated that any opportunity for reaeration, as in open water reaches and at weirs, is very beneficial in the oxygen economy of streams under winter, ice cover conditions.

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APPENDIX I - NOTATION

a = coefficient in weir equation
b = coefficient in weir equation
BOD = biochemical oxygen demand
BOD_5 = 5-day biochemical oxygen demand
cfs = cubic feet per second
DO = dissolved oxygen
D_o = DO deficit at beginning of reach, ppm
D_t = DO deficit at end of reach, ppm
fps = feet per second
FACT = factor for converting BOD_5 @ 20° C to BOD_5 @ 0° C
h = mean depth of river, feet
h_t = fall height at weir, feet
K_1 = first-order deoxygenation rate constant, to base e, day^{-1}
k_2 = first-order reaeration coefficient to base 10, day^{-1}
K_2 = first-order reaeration coefficient to base e, day^{-1}
L = ultimate first stage BOD, ppm
mi = mile
mm = millimeter
ppm = parts per million
Q = flow in cfs
r = deficit above weir/deficit below weir
ROD = residual oxygen demand, ppm per day
t = time, days
V = velocity in feet per second
y_t = BOD realized in time t
YREACH = first-order oxygen demand exerted in reach due to waste water conditions

POLLUTION - A BIOLOGICAL STUDY OF SOME RECEIVING WATERS IN HOKKAIDO

Matsunae Tsuda, Toshiharu Watanabe
and Kozo Tani

INTRODUCTION

The Ishikari River is about 365 km long and is the largest river in Hokkaido (Fig. 1). Its middle and lower reaches receive wastes from manufacturing industries, a pulp mill, and coal mines. Discoloration of the river water results from the pulp mill and mine waste while the river bed is covered by the coal mine silt.

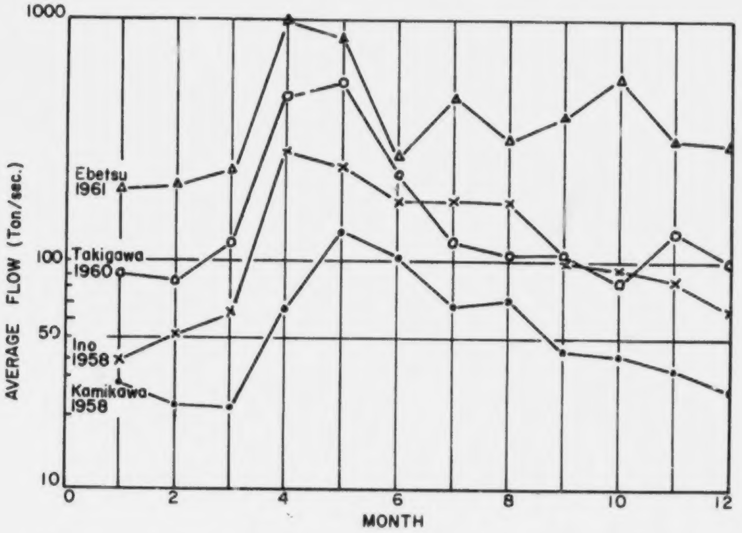

FIGURE 1 Average water flow of the Ishikari River at four stations

SOURCES OF POLLUTION

Upper Reach of the Ishikari

Near the city of Asahikawa (pop. 245,000) three tributaries - the Ushukubetsu River, the Chubetsu River and the Biei River - join the Ishikari River. The waste of the Kokusaku pulp mill is discharged into the Ushukubetsu River and that of the Godo Alcohol Company factory into the Chubetsu River. The effluent of the Kokusaku pulp mill has a BOD of 56.9 ppm and that of the Godo Alcohol a BOD of 3,998 ppm in January 1959 (Kubo and Kosaka, 1960).

TABLE 1

Water Temperature, pH, Transparency of the Stations Studied in the Ishikari River

		Water Temperature (° C)	pH	Transparency (cm)
A	Sounkyo	0.5	7.0	-
B	Motoshirakawa	2.8	6.8	-
C	Furukawa	2.3	6.8	-
D	Kinseibashi	2.8	6.6	46
E	Shinbashi	2.5	6.9	12
F	Kyokuseibashi	6.5	6.8	45
G	Ushubetsu R., near Kokusaku Pulp Mill Effluent	11.5	6.0	-
H	Ushubetsu R., Midoribashi	11.5	6.8	-
I	Ino	-0.5	6.8	18
J	Fukagawa	0	6.8	25
K	Sorachi R., Higashi-takegawa	1.0	6.8	11
L	Naie	1.0	6.7	11
M	Shimokawa	-	-	-
N	Ebetsu	1.0	6.8	5
O	Ishikari-ohashi	2.8	7.0	25
P	500 m downstream from paper mill effluent	7.0	6.9	6
Q	Toyohira R., Ganraibashi	0.5	7.0	30
R	Ishikari-cho	1.0	6.8	7

Middle Reach of the Ishikari

The Sorachi and the Yubari rivers receive coal mine effluents. The load of the suspended substances from the mines is estimated to be about 940 tons per day. Mine drainage and coal wash waste contain very fine coal dust which changes the water of the receiving river to a brown color and covers its bottom with silt. The influence of the coal mine wastes to the river organisms is physical - the deposited silt makes it difficult for organisms to survive. It ruins fish spawning beds and destroys flora and fauna. This form of pollution must be considered apart from the ordinary saprobic system classification.

Still another pollution source on the middle reach is an artificial fertilizer factory of the Toyo Koatsu Company located at Sunagawa. From this there discharges effluent containing much inorganic substances especially SO_4, the waste being heavily toxic to fish.

Organic Wastes Flowing in at the Lower Reach

A factory of the Kita-Nihon Paper Mill Company at Ebetsu discharges waste water of 45,000 tons per day, its BOD being 176 ppm (Tamura, Fuji, Yuki, 1961). Sewage treatment effluent and other sewage of the city of Sapporo (pop. 820,000), as well as industrial waste are discharged into a tributary, the Toyohira River.

BIOLOGICAL WATER QUALITY ASSESSMENT BY THE BOTTOM MICROORGANISMS

At 18 stations (from station A to station R) on the river, the microorganisms attached to stones, support piling and other suitable substrate, were collected and studied at the laboratory.

Table 2 shows the organisms found at stations A through R.

Upper Reach

The upper reach of the Ishikari was a clear mountain stream, with oligosaprobic waters, that were rich in algae and benthic fauna.

Station A (Sounkyo): A famous resort known for its hot springs.

Station B (Motoshirakawa): A small tributary.

Station C (Furukawa): 13 km downstream of Sounkyo.

At stations A, B and C, transparency was more than 45 cm. Many kinds of algae were found. Effluents from the hot-bath resort at Sounkyo have no important effect upon the river water. The benthic flora consisted of blue-green algae, green algae and diatoms. A chrysomonad, *Hydrurus foetidus*, occurred at Station A and *Bactrachospermum moniliforme* at B; these species are both cold-water kathorobic species. Among the diatoms, *Ceratoneis arcus*, *Fragilaria capucina* var. *lanceolata*, *Gomphonema olivaceum* were dominant. Stations A, B and C were typical oligosaprobic waters.

Asahikawa City and Surrounding Area

Station E (Shinbashi): This station was located on the left bank of the Ishikari River several hundred meters downward from the point where the Ushukubetsu River joins the main stream. This tributary receives waste of the Kokusaku Pulp Mill. Station E was heavily polluted and had a strong unpleasant odor. Here *Chromatium* colonies cover the stones. There was *Fusarium* growth which occurs frequently in waters with strong organic pollution and the presence of oxygen. Scheuring and Zehender (1962) write that *Fusarium* occurs more often in cellulose-containing wastes than the sewage bacteria *Sphaerotilus*. Practically no algae was found at this station. This station belonged to polysaprobic waters.

Station F (Kyokuseibashi): This station was on the right bank of the main stream, some hundreds of meters downstream from the junction of the tributary Ushukubetsu River, and 1 km downstream from the pulp mill, on the opposite side of the stream from the waste effluent, thus

TABLE 2

Benthic Microorganisms Found at the Stations in the Ishikari River

SPECIES	STATION																	
	A	B	C	D	E	F	G	H	I	J	K	L	M	N	O	P	Q	R
Bacteria																		
Zoogloea ramigera							+								‡		+	
Sphaerotilus natans									**	**	**	**	+		‡	+	+	
Chromatium sp.					‡													
Fungi																		
Alternaria sp.							‡						+					
Fusariam sp.					*				‡									
Blue-green algae																		
Chroococcus minutus	‡																‡	
Chlorogloea microcystoides																	*	
Dermocarpa flahaultii															*	+		
Xenococcus sp.			‡			+												
Oscillatoria formosa			**	+	+	‡												
Oscillatoria sp.																	+	
Phormidium uncinatum						‡		‡										
Phormidium spp.	‡											+						
Lyngbya sp.															+			
Homoeothrix janthina	‡		**	+														
Diatom																		
Melosira varians		+		+		+		+	+	+	+	‡		+	+	+		+
Cyclotella comta								+		+								
Tabellaria fenestrata		+																
Diatoma hiemale	‡										+							
Ceratoneis arcus	**	**	‡	+		+			+			+						
Fragilaria capucina var. *lanceolata*	‡	‡	*									‡						
Asterionella formosa											+							
Synedra ulna		‡	+	+				‡	+	+	+	+						+
S. ulna var. *Ramesi*			+	‡		+			+	+								+
S. capitellata f. striis			+	+		+				+								
Achnanthes linearis		+	+															
A. Biasolettiana		‡																
Cyrosigma acuminatum											+							
Navicula mutica											‡							
N. cryptocephala	*	+	+	+				+	+	+		+						+
N. exigua			+												‡			
Cymbella ventricosa	‡	+	+	+	‡		+		+	+								
C. turgidula var. *nipponica*			+	+		+												
C. tumida				+				+		+								
Gomphonema parvulum						+		+										
G. olivaceum			‡	‡		**				+	+							
Hantzschia amphioxys													+					
Nitzschia linearis											+							
N. palea		‡							+		‡	+	+	+		+	+	+
N. sigma											+					+		

TABLE 2 (cont'd)

SPECIES	A	B	C	D	E	F	G	H	I	J	K	L	M	N	O	P	Q	R
Green algae																		
Ulothrix subtilissima			+	‡	‡													
U. zonata	⧻	‡	+	+	+							+						
Hormidium Klebsii			+			+						+					‡	
Geminella crenulatocollis						+												
Stigeoclonium lubricum				*								+						
Rhizoclonium Hookeri		*																
Blubochaete sp.		⧻																
Oedogonium sp.		‡																
Closterium sp.		+																
Rhodophyceae																		
Batrachospermum moniliforme	‡																	
Chrysophyceae																		
Hydrurus foetidus	⧻																	
Ciliata																		
Vorticella campanula																		+
Carchesium polypinum										‡								⧻
Carchesium sp.															+			

** very rich, * rich, ⧻ common, ‡ rare, + very rare

unaffected. Many kinds of blue-green algae, green algae and diatoms were found. This station was oligosaprobic.

Station G (Ushubetsu River, Kokusaku Pulp): This was near the out-fall of the waste from the Kokusaku Pulp Mill on the right bank of the Ushukubetsu River. There was an unpleasant odor of factory waste which included the smell of hydrogen sulphide. The river water was dark brown in color. Stones were black-colored on their bottom surfaces due to sulphides. There were no aquatic insects. Growth on the stones consisted of *Zooglea* and *Alternaria* sp. (fungus), both of which are polysaprobic species which live in slow-running streams. This station belonged to polysaprobic waters.

Station H (Ushubetsu River, Midoribashi): This was also a station of the Ushukubetsu River, 1 km downstream from the paper mill. It was situated on the opposite side of the factory, but evidently influenced by the factory waste, as indicated by blackening of the bottom surfaces of the stones. Attached algae consist of blue-green algae (*Phormidium uncinatum*) and diatoms.

Station I (Ino) and Station J (Fukagawa): Both stations were on the left bank of the river. Influence of the paper mill was stronger here than on the other bank. At both stations there was an odor of pulp waste, brown water with reduced transparency, and the bottom surfaces of stones blackened. *Sphaerotilus* and *Fusarium* were found concurrently with diatoms. Both stations were α-mesosaprobic$^+$ (αm$^+$ means the class belonging to αm, but more strongly polluted than the average αm).

The Sorachi River and Sunagawa

In this reach the Ishikari River received the effluents of many coal mines. The wastes contained almost no organic compounds. Their effect on the river organisms was mostly the physical effect of siltation.

Station K (Higashi-takigawa): This station is at the Sorachi River, a tributary. Due to the coal mine waste, the water was brown-colored, with low transparency. The river bed was covered by silt and coal dust. Organisms were very scarce. Study of algae attached to stones covered by silt revealed that the flora consisted only of diatoms. (Table 2).

Station L (Naie): Located on the left bank of the main stream, this was 20 km downstream from the junction of the Sorachi River and 5 km downstream from the out-fall of the waste of the Toyo-Koatsu factory. Transparency was reduced and the water was brown in color.

Lower Reach and the Toyohira River

Station M (Shimokawa): This station was 40 km downstream from Station L. At both Station L and M, a few blue-green algae and diatoms were found. *Sphaerotilus* and fungus were also present, but in small quantity. Both stations are α-mesosaprobic.

Station N (Ebetsu): This station was 1 km downstream from the junction of the Yubari River, which carries a large amount of silt from coal mines into the Ishikari River. The river bed was covered by silt. Only two species of diatoms were represented here, and no benthic animals were found.

Station O (Ishikari-ohashi): This station was 1.5 km downstream from the junction of the Ebetsu River. There was much silt on the river bed, however transparency was recovered (Table 1). Micro-flora on stones consist of *Zooglea*, *Sphaerotilus*, blue-green algae and diatoms. The station was α-mesosaprobic*.

Station P: About 500 m downstream from the waste of the Kita-Nihon Paper Mill factory and also the out-fall of cooling water of a heat-engine plant; its water was brown-colored, with an unpleasant odor. *Sphaerotilus*, blue-green algae and diatoms were sparse. The station was polysaprobic.

Station Q (Ganraibashi): This station was on the Toyohira River, a tributary which flows through Sapporo City, the largest city in Hokkaido. The station was about 300 m downstream from the night soil treatment plant. Algae on stones consist mainly of blue-green algae, *Hormidiun* sp. (green algae), *Zooglea* and *Sphaerotilus* occurred too.

Station R (Ishikari-cho): This station was at Ishikari-cho, a town 5 km downstream from the mouth of the river. Water was brown in color, with transparency of 7 cm. Ciliates and a few diatoms were found. The diatoms consisted of such species as *Synedra ulna*, *Nitzschia palea*, *Melosira variens*, and *Navicula cryptocephala*. The station was α-mesosaprobic*.

Figure 2 shows the water quality map made on the basis of the biological assessment by the above-mentioned data on microorganisms.

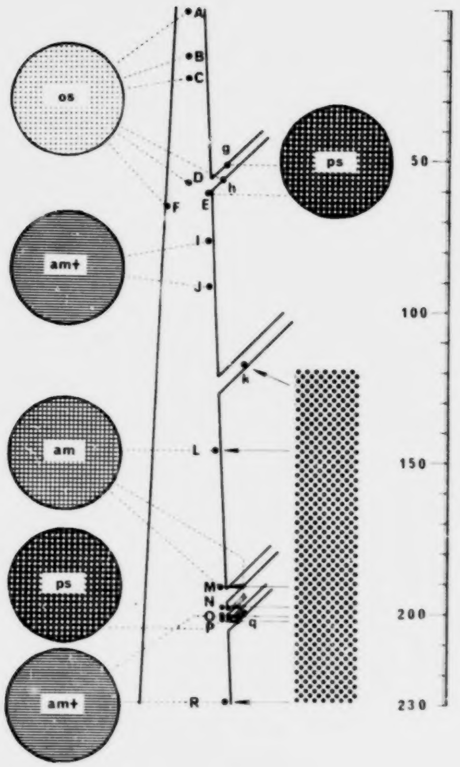

FIGURE 2 Water quality map of the Ishikari River by biological assessment

BENTHIC ANIMALS

Tables 3 and 4 show the benthic animals found at Stations A through R.

Upper Reach

The upper reach of the Ishikari was rich in benthic fauna.

Station A (Sounkyo): Large numbers of benthic species were found; the water was clear. The net-spinners coefficient (this equals the ratio of the biomass by weight of net-spinning caddis fly larva to the biomass of total benthic animals) was 82%, showing that the community was stable, i.e., at climax or near climax. Station A was oligosaprobic.

Station B (Motoshirakawa): Similar to Station A, oligosaprobic.

TABLE 3

Benthic Animals Found in Three Stations in the Upper Reach of the Ishikari River

SPECIES	STATION Sounkyo A	 Motoshirakawa B	 Furukawa C
Ephemeroptera			
Paraleptophelebia sp.	+		+
Ephemerella basalis			
Ephemerella rufa			
Ephemerella nigra	+		+
Ephemerella sp. nay	‡		
Ephemerella sp. EC	‡	‡	
Ephemerella sp. ER			‡
Baetis sp.	+		
Baetiella sp.			+
Isonychia japonica	+		
Epeorus uenoi	+		
Epeorus latifolium	+		
Rhithrogena sp.	+		
Cinygma sp.	+		‡
Plecoptera			
Nemoura sp.	*	+	
Amphinemoura sp.	‡		‡
Protonemura sp.	‡		‡
Capnia sp.	+		
Megarcys ochracea	+		
Isoperla towadensis	‡		‡
Hemiptera			
Notonecta trigutta		+	
Sigara nigroventralis			+
Megaloptera			
Sialis sp.		+	
Trichoptera			
Rhyacophila articulata	+		
Rhyacophila brevicephala			+
Rhyacophila sp. RG	‡		‡
Mystrophora sp. (Larva)	+		‡
Mystrophora sp. (Pupa)	‡		‡
Stenopsyche griseipennis	‡	+	‡
Arctopsyche sp. E	+		+
Hydropsyche ulmeri	*		‡
Hydropsyche sp. HB			+
Molanna falcata	+	+	
Neophylax ussuriensis	‡		
Glyphotaelius admorsus		+	
Stenophylax sp.		+	
Goera japonica		‡	+
Brachycentrus sp.	‡		
Micrasema sp.			+
Coleoptera			
Philorus ezoensis	+		
Tipula sp.		+	+
Antocha sp.	‡		‡
Spaniotoma sp.	‡		‡
Atherix sp.	‡		+
Turbellaria			
Dendrocoelopsis sp.			

TABLE 4

Benthic Animals of Stations E - R

SPECIES	STATION											
	E	F	H	I	J	K	L	M	O	P	Q	R
Ephemeroptera												
Ephemerella sp. EC		+										
Isonichia japonica Ulmey							+					
Plecoptera												
Isoperla towadesis Okamolo		‡										
Trichoptera												
Neoseverinia crassicornis Ulmey					+							
Coleoptera												
Ilybius apicalis Sharp, adult								+				+
Diptera												
Tipula sp.				+								
Antocha sp.		+										
Spaniotoma sp.		‡										
Tendipes sp.			+	‡	+	+					+	
Crustaceae												
Gammarus nipponensis Veno							‡		+			
Asellus ripponensis Nichols			‡	‡	‡		‡				+	
Oligochaeta												
Tubifex sp.			+	‡		+	‡			+	‡	⧻
Hirudinea												
Mimobdella sp.	+			‡	‡		+					

⧻ rich, ‡ common, + rare

Station C (Furukawa): Similar to Station A, oligosaprobic.

Asahikawa City and its Neighborhood

Station D (Kinseibashi): The water was clear; transparency was 46 cm. There were very few animals due to the devastation of the bottom by gravel-dredging. Kubo et al. (1961), at this location recorded larvae of some species of mayflies, stoneflies and caddisflies.

Station E (Shinbashi): This reach was extremely polluted by the waste of the Kokusaku Pulp Mill. *Mimobdella* sp. was the only animal found here. Station E was polysaprobic.

Station F (Kyokuseibashi): Some hundreds of meters downstream from the junction of the Ushukubetsu River, it was on the other side of the pulp mill, so there was little influence from the waste. *Ephemerella* sp. EC and *Isoperla towadensis*, which are clear-water species, were collected here. This station was oligosaprobic.

Station G (Ushubetsu River): This was located at the out-fall of the waste of the Kokusaku Pulp Mill. No benthic animals.

Station H (Ushubetsu River, Midoribashi): Station H was located 1 km downstream from the mill. *Tendipes* sp., *Asellus* sp., and *Tubifex* sp. were found, all of which are pollution-tolerant species. This station was β-mesosaprobic$^-$.

Station I (Ino): *Tendipes*, *Tipula*, *Asellus*, *Mimobdella*, and *Tubifex* were found. The station was α-mesosaprobic$^+$.

Station J (Fukagawa): *Tendipes*, *Asellus*, and *Mimobdella* were found. Besides clean-water insect species such as *Ephemerella* sp. EC, *Neoseverinia crassicornis* were found.

Lower Reach

Due to inflow of silt from the Sorachi River, a tributary, the river bed was covered by silt and coal dust. Almost no stones were found.

Station K (Higashi-takigawa): Station K was represented by *Tendipes* and *Tubifex*.

Station L (Naie): At Station L *Isonychia japonica*, *Gammarus*, *Asellus*, *Mimobdella* and *Tubifex* were found. Station L was β-mesosaprobic$^-$.

Station M (Shimokawa): The bottom at Station M was covered by silt and *Ilybius apicalis* was present.

Station N (Ebetsu): Silt and coal dust from coal mines characterized Station N; no benthic animals were found.

Station O (Ishikari-Ohashi): *Gammarus* sp. was found at Station O.

Station P: Station P was located 500 m downstream from the paper mill. The water was brown, and *Tubifex* and *Entosphenus japonicus* sp. were found. Lamprey were widely distributed in the middle and lower reaches of this river. They were found even in polluted waters, and seemed to be very tolerant of the pollution.

Station Q (Ganraibashi): Located on the Toyohira River downstream from Sapporo. The transparency of the water was 3 cm. *Tendipes*, *Asellus*, *Tubifex* and *Entosphenus japonicus* were found; *Tubifex* was very abundant. Station Q was α-mesosaprobic.

Station R (Ishikari-cho): The water transparency was 7 cm. The water color was brown and the bottom was covered with mud. *Tubifex* was present. Station R was α-mesosaprobic$^+$.

DISCUSSION

When we assess water quality biologically, we must consider two kinds of effects separately:

(a) The effect of organic pollution from municipal sewage and paper or pulp mills.

(b) The effect of silt from coal mines.

The normal saprobic system analysis can be used for (a) but not for (b). The effect of silt is recognized only by the following: when the effect is large, only few (or even no) species and only a few individuals will be found on the bottom of the river, and when the effect is small, more species and more individuals will be found. But in the case of organic pollution, we have indicators for every class of water quality.

In this river, the pollution by the pulp mills (and paper mill) is the most serious problem, especially the pollution by the factory of the Kokusaku Pulp Mill Company in Asahikawa City.

At Sunagawa, 90 km downstream from the factory the water color and BOD of the river water have recovered, but COD value is still high (40 ppm), and almost the same value continues to the estuary (Nakamura, 1961).

The importance of the dissolved organic materials from the peat zone in the drainage basin of the Ishikari should be also considered. These dissolved peat-originated substances are like the substances from the pulp wastes (lignin, cellulose, etc.) - very hard for bacteria to decompose. It must also be considered that the water temperature is low in most months of the year in this part of Japan. These are the reasons why the brown water color and high COD value hold in the whole middle and lower courses of the Ishikari River.

Suspended solids such as silt and sand from the coal mines are deposited on the bed of the Ishikari River, to which they are carried by the tributaries. The silt and sand will be carried farther downstream rolling on the bed. The bottom substances do not rest, since the bottom is not stable, thus the development of the benthic microorganisms, including bacteria, is very poor, indicating the reduced potentiality of natural purification of this river.

For the past two or three years the situation has improved. Many coal mines have stopped their operations owing to the increased cost-gain relationship. The pulp mill in Asahigawa City is moving part of its machinery to its other factory located nearer the sea.

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CHEMICAL EFFECTS OF SALMON DECOMPOSITION ON AQUATIC ECOSYSTEMS

David C. Brickell and John J. Goering

INTRODUCTION

The Pacific salmon (*Oncorhynchus* sp.) is a valuable source of food energy from the sea and its potential is not yet fully realized. The two most highly evolved species of the genus, *O. gorbuscha* and *O. keta*, are true marvels of efficiency from the anthropomorphic vista. Upon hatching in the gravels of coastal streams and rivers each spring, the salmon fry of these two species migrate immediately to the sea to harvest the productivity of the ocean waters. Their migrations extend over vast areas of the Central North Pacific.

After one year of residence in the ocean these salmon are compelled by instinct to return to their natal waters. This homeward migration results in the transfer of millions of pounds of organic matter, once widely distributed in the open ocean, to the coastal waters where salmon spawn. Thus, the vast primary productivity of the open ocean, which is currently too diffuse to harvest economically, is brought within man's grasp. This directed biomass transfer is sufficiently large to be considered a "biological current" originating in the Central North Pacific and terminating in the freshwater streams of the coast.

In addition to the large commercial harvest, of which a portion, about 20% by weight, is returned to the ocean as waste from processing plants, millions more salmon enter freshwater streams to spawn and inevitably to die and decompose. The fate and distribution of this organic matter has previously received little attention.

Although the decomposition of salmon carcasses in the marine environment is a natural phenomenon, the possibility that localized accumulations of seafood waste products, as might occur in the vicinity of processing plants, may stress the marine environment beyond its capacity to maintain stability must be considered.

Our interest in the fate of organic matter associated with seafood processing waste was stimulated by a Bering Sea cruise of the R/V ACONA, cruise number 066, in June 1968. In samples taken off the coast of Alaska in areas of significant seafood processing, namely in the area of Kodiak, Alaska and Unalaska, Alaska, we observed a water layer with extraordinarily high concentrations of ammonium. Near the processing plants toxic ammonium concentrations approaching 25 μg-atom NH_4^+-N/liter were observed while lower yet distinctly elevated concentrations were observed over tens of square miles of ocean in the area.

Other cruises of the R/V ACONA in the fjords of Southeast Alaska have revealed moderately high concentrations of ammonium at about 20-30 m. Evidence suggests that the origin of this ammonium is within the estuary rather than from the sea. Since the streams of Southeastern Alaska are abundant with salmon, the observed concentrations of ammonium may represent the results of carcass decomposition within the estuary.

To increase our knowledge of the biological and chemical effects of the decomposition of seafood material, we have initiated a study of the decomposition of salmon carcasses in a natural system in southeastern Alaska (i.e. Little Port Walter estuary). The Pacific salmon migrates through this estuary when returning to its natal stream to spawn. Following spawning, the fish die and the carcasses are eventually carried to the estuary where they sink to the bottom. During periods of low stream flow, the dead carcasses may remain in the stream itself until higher stream flows transport them to the estuary. In years of large escapements, the density of fish in the spawning stream can be very high. In the system chosen for our work, spawning densities greater than six fish per m^2 have been recorded, although at the time the current study was conducted the spawning density was slightly more than two fish per m^2. Since our system involves primarily pink salmon (*O. gorbuscha*), the average weight of the fish can be assumed to be 2-3 kilograms.

Thus, our study is concerned with the fate and distribution of some 75 metric tons of organic matter in the form of salmon carcasses in one small estuary in Southeastern Alaska. We are particularly interested in determining: (1) the effects of the salmon carcass decomposition on the nitrogen chemistry of the water in which the decomposition occurs; (2) the form and distribution of the organic matter which is returned to the marine system; and (3) the rate at which remineralization occurs. This paper presents the results of our initial investigations.

METHODS

The system selected for study included a pink salmon spawning stream, Sashin Creek, and its associated estuary, Little Port Walter, on Baranof Island, Southeastern Alaska. Another small estuary in the vicinity, Toledo Harbor, was used as a control for the estuarine studies as it did not support a salmon run.

Sashin Creek flows some 3,000 m from Sashin Lake to the Little Port Walter estuary. A high waterfall prevents further upstream migration of the salmon, and hence spawning is limited to the lower 1,200 m. The stream area above the water fall was used as a control for the stream studies since this area revealed seasonal variations in water chemistry but was not influenced by salmon. A permanent weir for counting fish entering the stream is located at high tide level.

The U.S. Bureau of Commercial Fisheries has maintained a research station at Little Port Walter since 1934 for the purpose of collecting information on the freshwater ecology of pink salmon. These studies have resulted in the definition of 3 distinct ecological areas of the stream: an upper, middle and lower area. To maintain integrity with existing data, we observed the same sampling boundaries in the stream. The total area of Sashin Creek available for spawning is about 13,600 m^2. The width of the stream varies between 12 and 24 m. Stream discharge data for Sashin Creek has been collected since 1951 and ranges from 8.0 to 700 cubic feet per second with a mean of 80 c.f.s.

The Little Port Walter estuary is small, consisting of an inner and outer bay with a constriction between the two. The distance from the mouth of Sashin Creek at high tide to the entrance to the outer bay is approximately 1.5 km, and the maximum width is about 0.4 km. The maximum depth at low tide is 44 m for the outer bay and 21 m for the inner bay. The connecting channel is 27 m wide and about 6 m deep at low tide. Tidal variations range to 4.6 m. A detailed description of the oceanography of Little Port Walter estuary is given by Powers (1962).

Toledo Harbor, about 3 km south of Little Port Walter, has dimensions approximately the same as the inner bay of Little Port Walter. The freshwater stream entering the harbor has a waterfall just above high tide level and does not therefore support a salmon run.

Stream Studies

Prior to the arrival of the salmon, surface water samples were collected weekly in Sashin Creek above the waterfall and at the lower end of each of the three designated ecological areas of the stream. Analyses for NO_2^--N, NO_3^--N, NH_4^+-N, dissolved organic -N and inorganic phosphate were performed. Nitrite was determined by the Griess method as applied to seawater by Strickland and Parsons (1965). Nitrate was determined by conversion to nitrite in a cadmium-mercury reduction column based on a method by Grasshoff (1964) and determined as nitrite. Ammonium was measured using the recent method of Solórzano (1969). This method is specific for ammonium and does not measure labile amino nitrogen. The ammonium method of Richards and Kletsch (1964) was used to obtain the Bering Sea ammonium values. This method is not specific for ammonium -N but measures a considerable fraction of labile amino nitrogen as well. Dissolved organic nitrogen was determined by converting it to nitrate by ultraviolet light oxidation in the presence of oxygen according to the method outlined by Armstrong, et al (1966). Reactive phosphorus was determined by the method of Murphy and Riley (1962). Samples were collected weekly beginning in August and continuing into November, about 2 months beyond spawning.

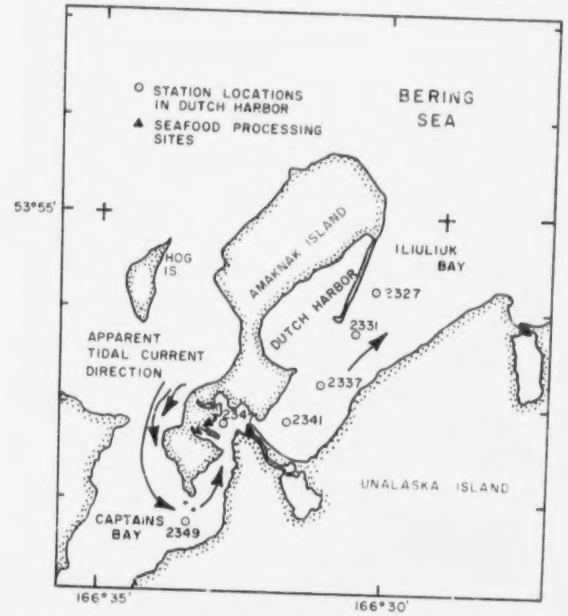
○ STATION LOCATIONS IN DUTCH HARBOR
▲ SEAFOOD PROCESSING SITES
BERING SEA
53°55'
HOG IS.
AMAKNAK ISLAND
DUTCH HARBOR
ILIULIUK BAY
2327
2331
2337
2341
234
APPARENT TIDAL CURRENT DIRECTION
UNALASKA ISLAND
CAPTAINS BAY
2349
166°35'
166°30'

FIGURE 1 Iliuliuk Bay, Unalaska Island, Alaska

TABLE 1

Chemical and Physical Characteristics of Water in Iliuliuk Bay at Stations 2327 - 2349 Near Seafood Processing Sites and at Station 2351 further Seaward

Station Number and Location	Depth in Meters	Temp. C	Salinity °/°°	Oxygen ppm	PO_4^{-3}-P µg-atom/liter	NH_4^+-N	NO_2^--N µg-atom/liter	NO_3^--N µg-atom/liter
2349	0	5.70	32.051	7.91	1.08	0.4	0.17	5.8
53°51.8'N 166°33.3'W	5	5.63	32.005	7.97	1.10	0.4	0.16	6.0
	10	5.01	32.237	7.67	1.35	0.8	0.19	7.8
	15	4.81	32.266	6.90	1.44	0.8	0.19	11.2
	20	4.34	32.294	7.14	1.69	1.0	0.20	13.1
	30	3.70	32.360	6.84	2.12	0.9	0.20	18.3
	40	3.30	32.343	5.60	2.45	1.0	0.22	20.9
2347	0	7.35	24.906	6.78	1.22	5.2	0.19	0.9
53°52.8'N 166°32.7'W	5	5.52	32.171	7.17	1.78	8.1	0.26	5.1
	10	5.20	32.223	7.39	1.50	5.7	0.23	6.6
2341	0	6.71	30.782	7.98	0.51	1.8	0.06	0.2
53°52.8'N 166°31.5'W	5	5.98	31.667	8.05	0.87	3.4	0.11	2.2
	10	5.39	32.202	7.82	1.21	4.6	0.15	4.7
	15	5.03	32.366	7.35	1.47	5.9	0.16	6.4
	25	3.83	32.373	4.30	5.03	23.8	0.33	9.2
2337	0	5.95	31.788	8.07	0.84	2.4	0.12	3.0
53°53.2'N 166°31.1'W	5	5.86	31.820	8.08	0.88	2.3	0.14	3.0
	10	5.47	32.265	8.10	0.98	2.2	0.12	4.5
	15	5.19	32.313	7.83	1.20	4.9	0.15	5.6
	25	3.79	32.381	4.84	4.43	19.9	0.33	9.6
2331	0	6.70	30.781	7.88	0.48	1.7	0.06	0.5
53°53.7'N 166°30.6'W	5	5.87	32.109	8.34	0.83	2.0	0.12	3.4
	10	4.92	32.304	7.51	1.42	6.4	0.17	6.7
	15	4.64	32.347	7.36	1.80	9.3	0.21	7.7
	25	4.11	32.362	5.51	3.37	18.4	0.28	9.9
2327	0	6.02	31.777	7.93	0.70	2.7	0.09	2.3
53°54.2'N 166°30.2'W	5	5.68	32.146	7.97	0.96	3.2	0.10	3.2
	10	5.22	32.376	7.59	1.13	5.0	0.13	5.7
	15	-	32.391	7.28	1.42	6.5	0.16	7.0
2351	0	4.87	-	-	1.30	3.4	0.16	12.0
54°01.3'N 166°04.5'W	5	4.88	32.431	-	1.29	3.6	0.17	11.7
	10	4.88	32.408	-	1.25	3.2	0.18	11.5
	20	4.87	32.502	-	1.26	3.3	0.17	11.8
	30	4.88	32.412	-	1.25	3.2	0.17	11.8
	50	4.87	32.523	-	1.26	3.2	0.17	12.3

Estuary Studies

In the estuary, weekly samples were collected at selected depths in both Little Port Walter and Toledo Harbor. The same chemical analyses and procedures were employed as were used in the stream studies.

RESULTS

Iliuliuk Bay and Bering Sea Studies

The locations of sampling stations and seafood processing plants on Amaknak and Unalaska Islands, which are located near Unimak Pass in the Aleutian chain, are shown in Figure 1. The oceanographic parameters observed in the area on cruise 066 indicated that during our sampling period in June 1968 the tidal current entered the area from the northwest and exited toward the northeast. The temperature-salinity-oxygen characteristics in the Captain's Bay area where station 2349 is located suggested an open ocean environment little affected by the waste of the processing plants, while those stations to the northeast of the processing plants revealed rather marked chemical alteration. Table 1 presents the chemical and physical parameters obtained at all stations. Station 2349, indicative of the water entering the area of seafood processing, had a normal open ocean ammonium concentration of approximately 1μg-atom NH_4^+-N/liter at 25 m while the ammonium concentration at 25 m at station 2341 was 23.8μg-atom NH_4^+-N/liter, which is an extraordinarily high concentration for sea water. Station 2351, further seaward, exhibited the chemical characteristics which result after the water from Iliuliuk Bay has mixed with Bering Sea water. The ammonium values obtained in this study of Iliuliuk Bay were determined using the method of Richards and Kletsch (1964). This method does not distinguish ammonium -N from labile amino -N. Because of the highly organic nature of the seafood waste a considerable fraction of the nitrogen reported here as ammonium -N could be amino -N.

Figure 2 presents a plot of the ammonium and oxygen concentrations with depths at stations sampled in Iliuliuk Bay. The results show a drastic increase in the ammonium concentration of the

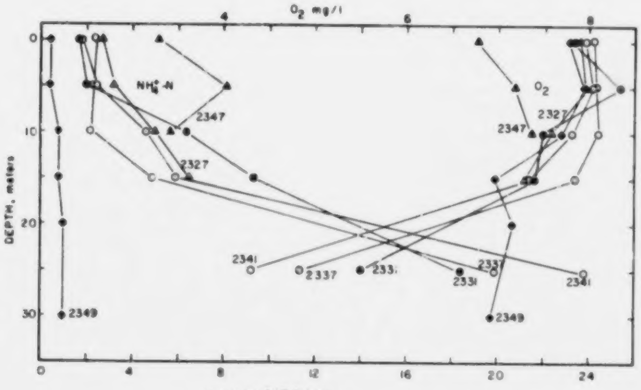

FIGURE 2 NH_4^+-N and oxygen at various stations and depths in Captain's Bay and Iliuliuk Bay waters while in residence in Iliuliuk Bay. Whereas that water entering the bay from Captains Bay has an ammonium concentration of approximately 1μg-atom NH_4^+-N/liter. The concentration of ammonium in the bottom waters within the bay increases to almost 25μg-atom NH_4^+-N/liter. Likewise, the oxygen concentration of the bottom waters is lowered. The ammonium concentrations are high at intermediate depths, 5 - 10 μg-atom NH_4^+-N/liter at 15 m, but the most significant increase in ammonium and decrease in oxygen occurs between 15 and 25 m. Since the 25 m sample is near the bottom, the substantial increase in ammonium at this depth probably results from decomposition of organic matter which has accumulated on the bottom.

The data for station 2351 indicated a well mixed water column with the ammonium distributed rather uniformly through the column from the surface to 50 meters. Other seaward stations displayed decreasing but elevated ammonium concentrations, suggesting that ammonium originating in Iliuliuk Bay has an influence on the nitrogen economy of the surrounding ocean.

Stream Studies

After evaluating the data from the Iliuliuk Bay and Bering Sea study we decided to examine a natural phenomenon which results in the accumulation of organic matter in the marine environment similar to the situation observed in the vicinity of seafood processing sites. Salmon inevitably die following spawning. This results in the accumulation of salmon carcasses both in the freshwater stream where spawning occurs and in the receiving estuary. This natural system was selected for study because it represented a cyclic phenomenon which could be followed from beginning to end with little external influence.

About 30,000 pink salmon spawned in the 1200 m of Sashin Creek in 1969. After spawning and death many of these carcasses remained in the stream for a period of time before being washed into the estuary. Some of these were scavenged, but the majority of the carcasses were deposited on the bottom of the Little Port Walter estuary. In following the chemistry of salmon carcass decomposition we were particularly interested in the nitrogen chemistry. Since fish flesh contains much protein we felt that the various forms of nitrogen would be excellent indices of the rates of biological decomposition of salmon carcasses.

Surface water samples were collected and analyzed weekly from four different sites in the stream; above the waterfall which prevented further upstream migration of the spawners, Area O, and at the lower boundary of each of the three defined ecological areas. These boundaries were as follows: Area I, waterfall - 430 m downstream; Area II, 430 - 730 m downstream; and Area III, 700 - 1200 m downstream. The first 300 m of stream below the waterfall is not favorable for spawning and hence only a few carcasses are deposited here.

Since the water at each sampling area downstream had been exposed to more carcasses, we expected a systematic increase in the concentrations of metabolites resulting from carcass decomposition as we moved progressively downstream.

Water samples collected prior to entrance of the fish into the stream showed little variation in water chemistry from above the falls to the mouth of the stream. On August 31, the NH_4^+-N concentration above the fall was 0.32 μg-atom N/liter while at the stream mouth the concentration was 0.46 μg-atom N/liter. Similarly, the dissolved organic nitrogen concentration above the falls on that date was 3.3 μg-atom N/liter, while at the stream mouth an organic nitrogen concentration of 3.8 μg-atom N/liter was observed.

Spawning activity commenced in late August and by early September dead carcasses began to appear in the stream. The first noticeable chemical effects of decomposition were observed in the stream on September 14. On this date, the ammonium concentration above the falls was 0.40 µg-atom N/liter while at 1200 m downstream the concentration increased to 1.73 µg atom N/liter. Likewise the dissolved organic nitrogen concentration increased from 4.0 µg-atom N/liter to 7.4 µg-atom N/liter during the traverse. As the number of carcasses increased, the downstream increase in ammonium nitrogen and dissolved organic nitrogen became greater. On September 29 we observed the greatest difference in stream chemistry between the control area and the lower end of the stream. The ammonium concentration increased from 1.66 µg-atom N/liter above the falls to 7.80 µg-atom N/liter at 1200 m downstream. Dissolved organic nitrogen increased from 5.7 µg-atom N/liter to 17.7 µg-atom N/liter. The concentration of NH_4^+-N and dissolved organic -N observed at each sampling site and date are presented in Table 2 and in Figures 3 and 4.

TABLE 2

The Concentrations of Ammonium -N and Dissolved Organic -N (D.O.N.) Observed in each Sampling Area of Sashin Creek

Concentrations are Expressed as µg-atom-N/liter

	Area 0		Area I		Area II		Area III	
Date	NH_4^+-N	DON	NH_4^+-N	DON	NH_4^+-N	DON	NH_4^+-N	DON
8/24/69	0.20	3.04	0.60	3.18	0.62	3.27	0.68	3.67
8/31/69	0.32	3.29	0.38	3.39	0.38	3.49	0.46	3.84
9/7/69	0.32	3.38	0.64	3.64	0.92	4.84	0.98	5.39
9/14/69	0.40	4.04	0.33	5.31	1.40	6.10	1.73	7.36
9/22/69	1.31	5.83	1.44	6.44	3.22	8.22	4.17	9.71
9/29/69	1.66	5.71	4.59	7.38	6.55	12.40	7.80	17.67
10/8/69	1.56	5.23	4.48	6.72	6.62	10.52	7.66	16.8
10/16/69	1.44	5.22	4.04	6.11	6.72	8.43	7.02	10.32
10/28/69	1.63	4.84	2.46	6.02	3.88	7.02	5.63	8.64

Estuary Studies

Water samples in the Little Port Walter estuary and in the control estuary, Toledo Harbor, were collected weekly at 1, 4, 8 and 12 m. Dissolved organic nitrogen proved to be the most valuable index of the salmon carcass decomposition phenomenon. The dissolved organic nitrogen concentrations observed in Little Port Walter and Toledo Harbor appear in Table 3. Figure 5 compares graphically the dissolved organic nitrogen of the surface waters of the two estuaries. Figure 6 is a comparison of their organic nitrogen concentrations at 12 m.

The first effects of salmon carcass decomposition on the water in Little Port Walter estuary, as evidenced by increases in dissolved organic nitrogen, were not observed until about 15 days after

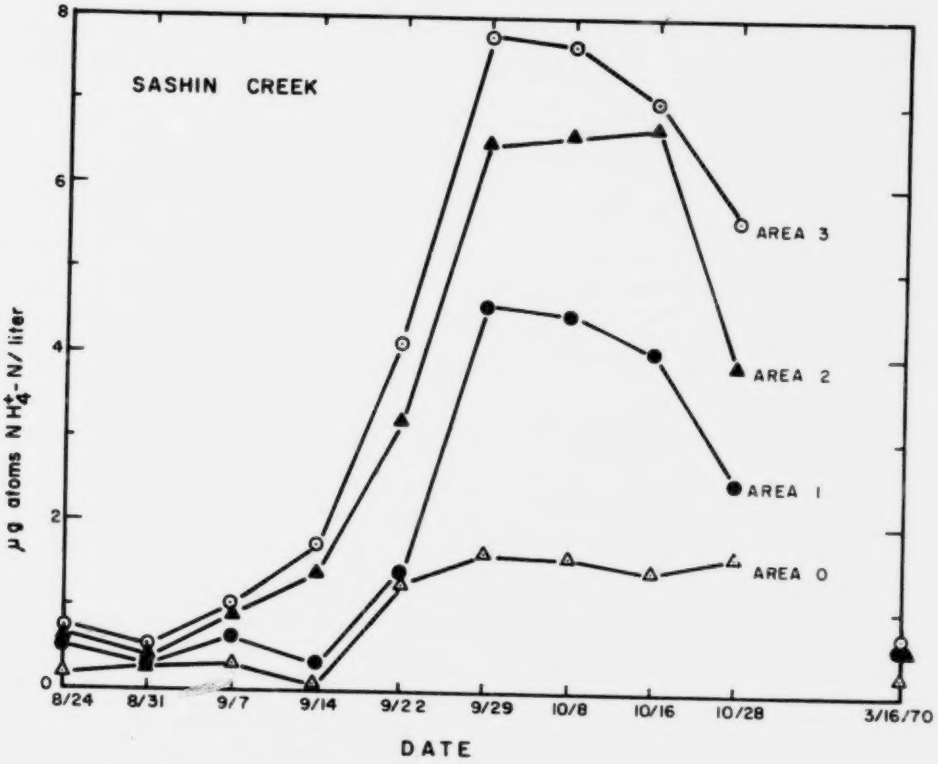

FIGURE 3 Concentrations of NH_4^+-N in Sashin Creek surface water

increases were first observed in the stream. Heavy rains in late September resulted in flushing most of the carcasses from the stream into the estuary where they sank to the bottom. The dissolved organic nitrogen concentration in the surface water of Little Port Walter estuary began to increase on September 22 and reached a peak of 10.3 μg-atom N/liter on October 28 (Fig. 5) while the surface concentration in Toledo Harbor remained almost steady at about 5.4 μg-atom N/liter throughout this period.

The data on ammonium concentrations for the two estuaries did not reflect the immediate effect of carcass decomposition. The concentrations in fact declined during the fall months from a summer high. The ammonium concentrations observed at 12 m depth in the two estuaries are presented in Figure 7.

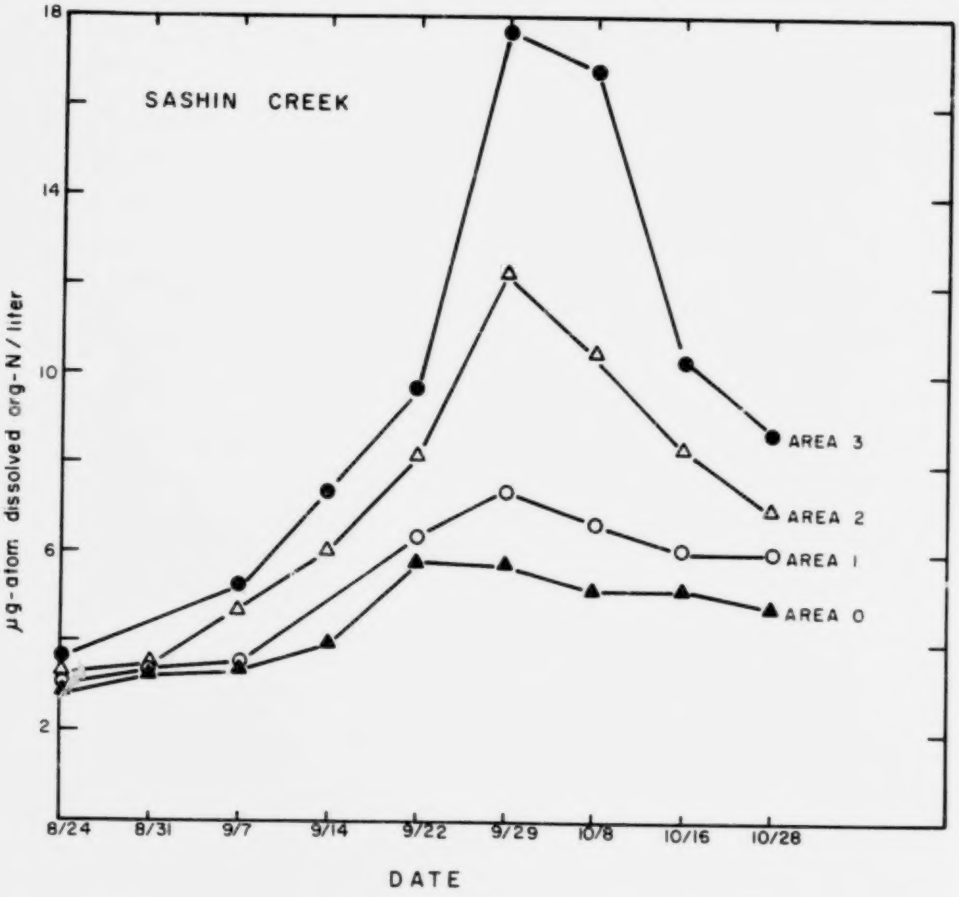

FIGURE 4 Concentrations of dissolved organic -N in Sashin Creek surface water

TABLE 3

Dissolved Organic Nitrogen Concentrations Observed in Little Port Walter and Toledo Harbor Estuaries in Summer and Fall of 1969

Concentrations Expressed as μg-atom-N/liter

Depth m	8/24	8/31	9/7	9/14	9/22	9/29	10/8	10/16	10/28	11/14
Little Port Walter										
1	2.42	2.40	3.96	3.27	3.78	5.46	6.62	9.20	10.33	5.62
4	2.64	1.46	3.03	3.40	1.48	3.36	7.52	4.86	10.83	4.83
8	2.42	1.32	3.02	3.18	1.09	3.50	2.55	11.61	9.11	7.68
12	2.56	3.90	3.38	3.65	1.80	6.53	8.92	16.26	14.26	12.43
Toledo Harbor										
1	2.07	1.52	1.45	1.34	1.23	5.24	5.48	5.42	5.32	5.02
4	1.87	0.65	1.69	1.41	1.20	5.60	5.66	5.63	4.27	4.22
8	3.21	1.60	1.98	1.88	1.32	4.98	3.45	4.99	5.39	4.87
12	2.46	3.09	2.14	2.35	1.92	4.93	4.63	5.02	5.42	5.20

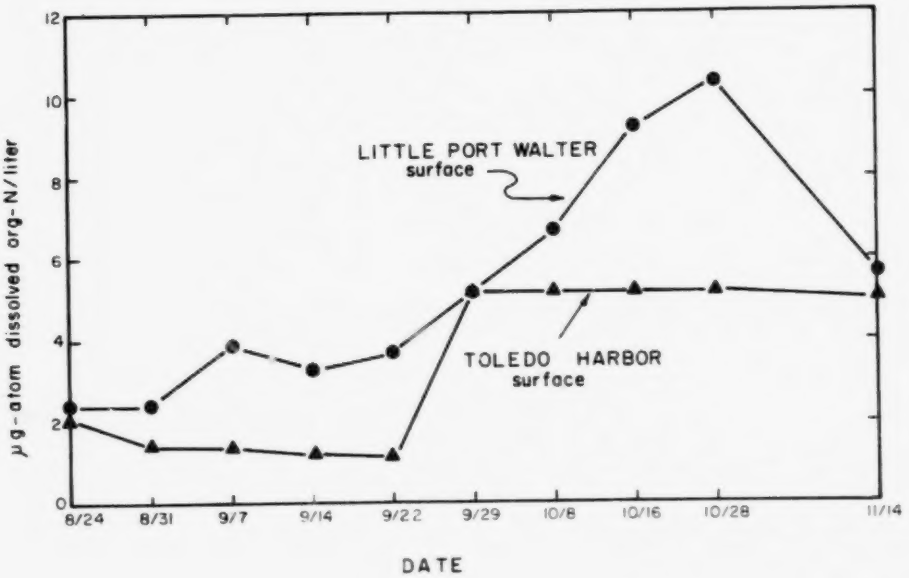

FIGURE 5 Concentrations of dissolved organic -N in surface water in Little Port Walter and Toledo Harbor estuaries

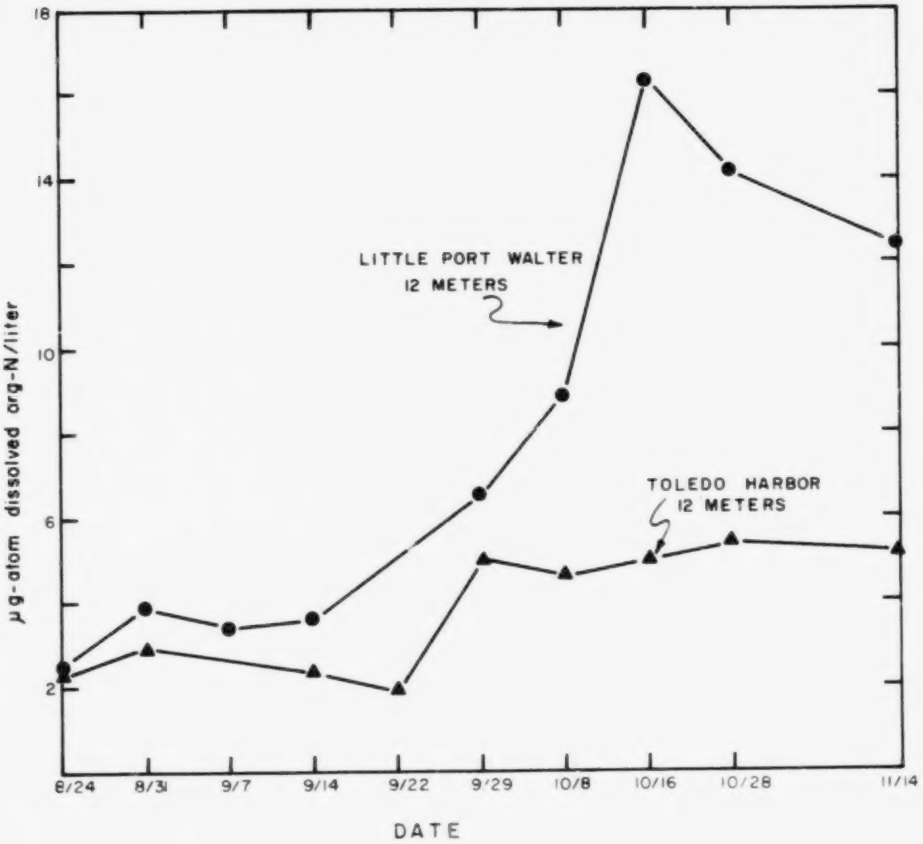

FIGURE 6 Concentrations of dissolved organic -N in 12m water in Little Port Walter and Toledo Harbor estuaries

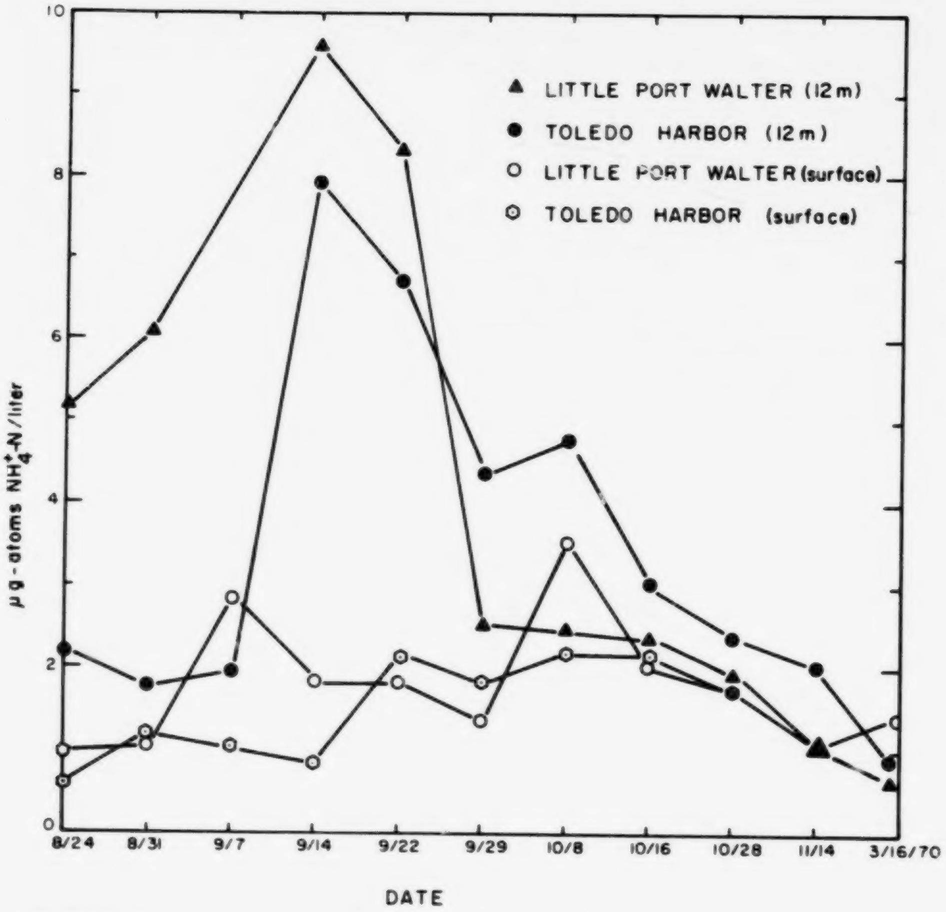

FIGURE 7 Concentrations of ammonium -N at 12m in Little Port Walter and Toledo Harbor estuaries

DISCUSSION

The surprisingly large concentrations of NH_4^+-N and the oxygen depletions observed in Iliuliuk Bay indicate that the decomposition of seafood material in a restricted body of water can influence the nitrogen chemistry of the estuarine environment. The organic waste matter resulting from seafood processing plants located on Iliuliuk Bay is emptied into the estuary rather continuously. Thus the highest observed concentrations of NH_4^+-N at station 2341, 23.8 μg-atom N/liter may well represent the remineralization of organic material that has accumulated over a long period of time. It would appear that such extremely high concentrations of ammonium are potentially toxic to fish (Ball, 1967) and to other forms of life. Other effects of decomposition such as oxygen depletion are also probably detrimental to the organisms residing in the bay.

It is important to determine whether elevated concentrations of ammonium as were observed in Iliuliuk Bay occur naturally in the marine environment. It appeared from the knowledge available on the nitrogen chemistry of the sea that it would be unlikely that such high concentrations of NH_4^+-N would occur naturally, and that the observed situation was therefore the result of industrial stress on the environment. We felt that the phenomenon of salmon carcass decomposition most closely duplicated the industrial situation, and we were therefore interested in determining what level of nitrogen compounds were present in systems where salmon were decomposing.

It is difficult to compare the Richards and Kletsch (1964) ammonium values for Iliuliuk Bay to those obtained by the Solórzano (1969) technique in Little Port Walter, since it is uncertain what fraction of the organic nitrogen was recorded by the Richards and Kletsch method as ammonium. Our investigations in Little Port Walter suggest that dissolved organic nitrogen is the initial decomposition product and organic nitrogen (i.e., labile amino-N) may have contributed heavily to the ammonium measured in Iliuliuk Bay. If the ammonium -N and dissolved organic -N for Little Port Walter are summed, a combined dissolved organic -N · ammonium -N concentration of 18.2 μg-atom N/liter occurred near the bottom on 10/16/69. This approximates the value of 23.8 μg-atom of NH_4^+-N/liter observed in Iliuliuk Bay and indicates that high concentrations of ammonium and organic nitrogen are not restricted to waters receiving waste from seafood processing.

The ammonium and dissolved organic nitrogen concentrations observed in Sashin Creek indicate that decomposing salmon carcasses have a significant influence on stream chemistry. It has been observed that survival of salmon eggs in the Sashin Creek spawning beds was significantly lower in the lower area of the stream, Area III, than in the upper areas. Several reasons have been presented by McNeil (personal communication) to explain this phenomenon. The chemical parameter normally associated with survival of eggs in the spawning bed is dissolved oxygen, but we feel that ammonium or other products of carcass decomposition might have a significant influence on egg survival, especially when stream flow is low.

The large dissolved organic nitrogen concentrations observed in the surface waters of Little Port Walter undoubtedly reflected the decomposition of salmon carcasses occurring in the stream and in the intertidal zone. The bottom of the estuary received most of the carcasses and it is here that final decomposition occurs. Sediment samples of the Little Port Walter estuary bottom showed a gelatinous quality even for samples taken prior to the arrival of the fish, suggesting that the sediment may act as a nutrient sink. This gelatinous nature probably results from the long term accumulation of fish carcasses. All sediment samples from Toledo Harbor were granular.

The apparent increase in the concentration of dissolved organic nitrogen observed in the control estuary, Toledo Harbor, between September 22 and September 29 was probably the reflection of heavy rains, wind and heavy cloud cover which resulted both in high run-off from the land and death of plankton in the water column.

It is evident that this brief study is only a beginning in our understanding of the fate of organic matter deposited, naturally or by industry, in the marine environment. Our attempts to follow the decomposition of salmon carcasses employing nitrogen chemistry was rewarding in that effects were clearly demonstrable. Since nitrogen comprises only a small proportion of the organic matter, and since remineralization rates are slow, it is evident that more sophisticated techniques must be utilized to obtain a clear understanding of the process. Possibly, remineralization of the organic matter resulting from carcass decomposition does not occur within the confined body of water where the carcasses are deposited. The early stages of decomposition might result in the production of soluble complex organic compounds which are transported from the local area before remineralization occurs.

ACKNOWLEDGMENTS

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PHOSPHORUS BINDING MECHANISMS DURING SELF-PURIFICATION OF POLLUTED LAKES

Jan Werner

INTRODUCTION

The discharge of sewage into lakes and rivers gives rise to different and fundamental changes. Among the various parameters of importance that can influence the ecosystems, the following three will be discussed:

1. The oxygen consuming substances
2. The phosphorus compounds
3. The alkalinity

The increases in phosphorus concentration and alkalinity are two factors which are supposed to affect the primary production of a limnic system directly and positively (Christie, 1968; Mortimer 1941, 1942). The increased production will, in combination with the external input of organic substances, sometimes cause a complete depletion of the oxygen in the bottom waters during periods of stratification. The sediments will be especially exposed to such conditions, as the organic particles will settle to the bottom and form part of the upper surface of the sediment.

In this connection the exchange reactions between water and sediments seem to be of considerable importance. The sediments act as large buffers and storing places for various types of solid materials. Exchange mechanisms will cause a shift of certain substances between the solid and the dissolved states. Phosphorus is one of the substances that will be affected by the exchange mechanisms. The fundamental experiments of Mortimer (1941, 1942) corroborate the general experience that when oxygen is available to the sediments the transfer of phosphorus from a solid to a dissolved state will be very limited. There will on the other hand be an increase of soluble phosphorus in the sediment system when oxygen is depleted.

CIRCULATION OF PHOSPHORUS AND ITS ACCUMULATION IN THE SEDIMENT SYSTEM

During summer, process 2 tends to dominate over process 3 with an enrichment of phosphorus in the hypolimnion and in the sediment system as a consequence. During winter, process 2 will again dominate over process 3, especially during the period following the formation of an ice cover. The spring and fall turnovers will on the other hand cause an even distribution of the phosphorus throughout the whole water mass, i.e. process 3 will be favored over process 2.

The sediment material with which phosphorus is associated will be broken down successively to more simple substances by biochemical processes. Whether phosphorus is bound to organic molecules or not, orthophosphate will be the principal end product. To reduce the amount of phosphorus circulating within a body of water, mechanisms causing an accumulation in the sediment must be made to operate. There are in the first instance two principally different mechanisms possible.

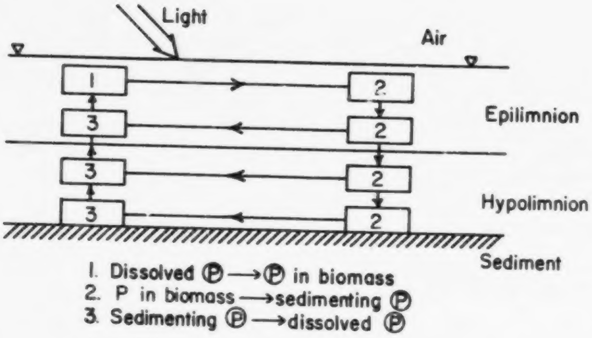

Light
Air
Epilimnion
Hypolimnion
Sediment
1
2
3
1. Dissolved Ⓟ ⟶ Ⓟ in biomass
2. P in biomass ⟶ sedimenting Ⓟ
3. Sedimenting Ⓟ ⟶ dissolved Ⓟ

FIGURE 1 Circulation of phosphorus and its accumulation in the sediment system

1. An essential part of the settling organic phosphorus compounds will decompose slowly. The accumulation according to this mechanism will reflect a dynamic equilibrium determined by the quantity settling, the rate of decomposition, and finally the quantity released from the sediment.

2. Phosphorus compounds, especially orthophosphate, will be bound to undefined inorganic solids in the sediment system. There will be set up a chemical equilibrium between absorbed phosphorus and phosphorus dissolved in the interstitial water and in the water above and near the sediment surface.

BINDING OF PHOSPHATE TO FERRIC IRON

In a lake with low primary production and with a limited input of oxygen consuming substances, oxygen will normally be available in the bottom waters. In such a system any ferrous iron will be oxidized to ferric iron, provided it is not present as a very stable chelate. Ferric iron forms stable and partially soluble molecular aggregates under the prevailing conditions. These aggregates may have different anions incorporated, of which the following three are of special interest: phosphate, hydroxyl and humic substances. Iron is normally present in a large excess compared to phosphorus. The conditions in the sediment may be illustrated as in Figures 2 and 3. If ferric hydroxide in the sediment is in excess the solubility of the ferric iron will be determined by the solubility product of ferric hydroxide. We may assume that the hydroxide exists in various loosely crystallized modifications. The lack of defined crystallinity makes it fairly reactive. The reported stability constants for ferric hydroxide range from 10^{-37} to 10^{-44} (Feitknecht, 1959). This reflects that the stability of ferric hydroxide will depend on the conditions at which it is formed and also its age, etc. This raises difficulty when a value relevant to the conditions in a sediment system is to be selected. It seems reasonable, however, that the stability constant would be closer to 10^{-37} than to 10^{-44}, i.e. the hydroxide should be comparatively reactive.

If ferric hydroxide is in excess compared to ferric phosphate, the concentration of soluble phosphate in bottom waters and its dependence on pH may be illustrated as in Figure 4. The temperature of the bottom water has been assumed to be about 5° C. The ionic product of water, K_w, will then be $10^{-14.7}$. The derived figure should be considered as an approximation as the

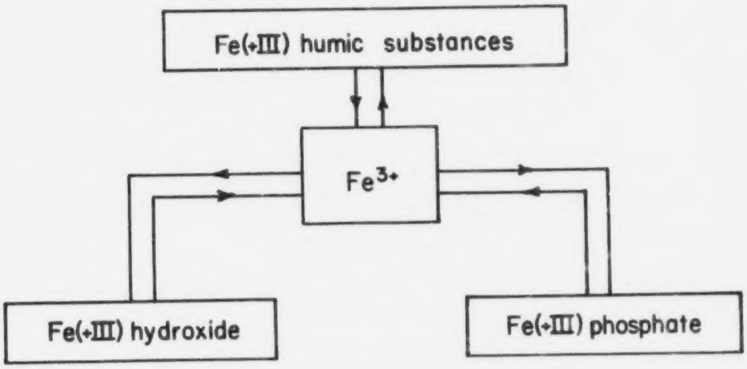

Fe(+III) humic substances
Fe3+
Fe(+III) hydroxide
Fe(+III) phosphate

FIGURE 2 Binding of phosphate to ferric iron

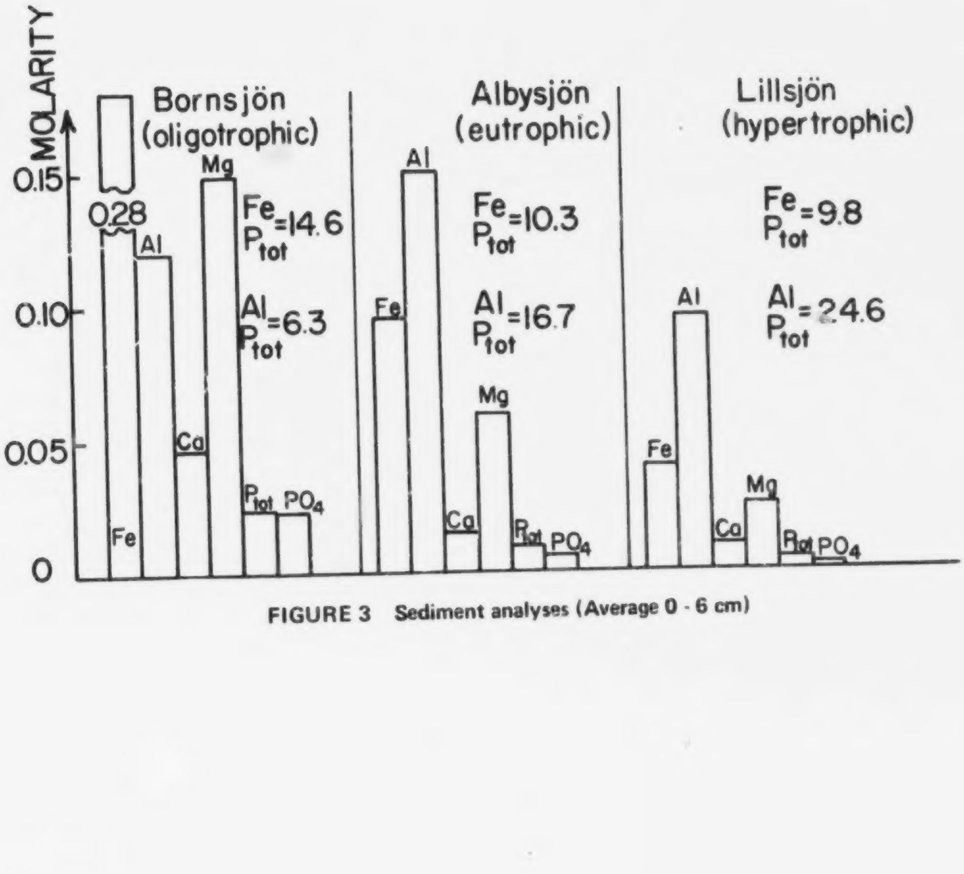


FIGURE 3 Sediment analyses (Average 0 - 6 cm)

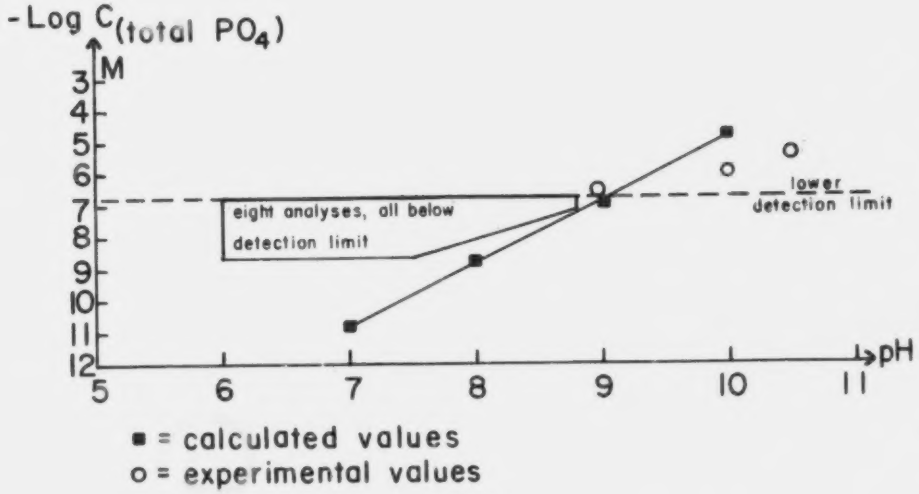


FIGURE 4

corrections of certain data for temperature have not been possible. The constants used for calculation of the concentrations of different ionic species of phosphate have been taken from Egan et al. (1961). The figure also includes a curve derived from an actual experiment. It appears that the experimental curve fits fairly well to the theoretically derived curve when 10^{-37} is selected as stability constant for ferric hydroxide under the discussed conditions.

The figure indicates that:

a) ferric hydroxide in a sediment system is of a relatively reactive type, and

b) the concentration of soluble phosphate is highly dependent on pH, being near or below the detection limit under the condition expected in the system.

BINDING OF PHOSPHATE TO Ca^{2+}, Mg^{2+} AND Al^{3+}

Phosphate ions can be bound to ferric iron, aluminum, calcium and magnesium. Precipitation of phosphate by calcium or magnesium takes place only at higher concentrations than those discussed. At the pH which prevails in the bottom waters, precipitate of calcium or magnesium phosphate takes place only to a limited extent.

Aluminum will on the other hand react with phosphate in a way analogous to ferric iron. The resulting complexes have approximately the same stabilities and the same dependances on pH. As a consequence, any discussion of the role of iron as a phosphate-fixing agent in sediments must consider the possible competition between aluminum and ferric iron. This is of special importance under conditions when ferric iron might be reduced to the ferrous state.

The total amount of iron and aluminum present at the surface of the sediment exceeds the phosphorus content on a molar basis more than tenfold. The fact that in spite of this a complete fixation of phosphate in the sediment system does not take place will need careful analysis.

REDUCTION OF FERRIC IRON AND OXYGEN

The reduction of ferric iron must be considered with reference to the following:

a) Fe^{3+} is an important oxidizing agent and a redox buffer at the surface of the sediment.

b) When ferric iron is reduced to ferrous iron, it loses its properties to bind phosphate, hydroxyl ions and humic substances. A physico-chemical analysis of the reaction

$$Fe^{3+} + e^- \rightleftarrows Fe^{2+} \qquad E^\circ = 770 \text{ mV at } 25^\circ C \text{ (Schumb et al., 1937)}$$

will give a redox value according to Nernst's equation

$$E_{Fe^{3+}/Fe^{2+}} = 770 + 55 \log \frac{(Fe^{3+})}{(Fe^{2+})} (55 \text{ mV at } 5^\circ C)$$

The Fe^{3+}-concentration will be determined by the reaction

$$Fe^{3+} + 3\,OH^- \rightleftarrows Fe\,(OH)_3\ (s)$$

Fe^{3+} will vary according to the expression

$$Fe^{3+} = \frac{10^{-37.0}}{(OH^-)^3}$$

With the ionic production of water being $10^{-14.7}$ at 5° C, the ferric ion concentration will vary with pH according to

$$(Fe^{3+}) = 10^{-37.0} + 3 \cdot 14.7 - 3\,pH = 10^{+8.6 - 3\,pH}$$

If the concentration of Fe^{2+} is assumed to be 10^{-5} M (0.6 ppm) and if we use the expressions for Fe^{3+} and Fe^{2+} given by Nernst's equation the following expression will result

$$E_{Fe^{3+}/Fe^{2+}} = 770 + 55 \log \frac{10^{+8.6 - 3\,pH}}{10^{-5}}$$

$E_{Fe^{3+}/Fe^{2+}}$ as a function of pH is given in Figure 5.

Oxygen may be reduced in the following way:

$$1/2\,O_2 + H_2O + 2\,e^- \rightleftarrows 2\,OH^- \qquad E^\circ = 401 \text{ mV at } 25^\circ C \text{ (Latimer, 1952)}$$

The redox potential varies according to

$$E_{O_2/OH^-} = 401 + \frac{55}{2} \log \frac{P_{O_2}^{1/2}}{(OH^-)^2}$$

The O_2/OH^- system will be the determinant for the oxidation or reduction of iron.

P_{O_2} is given as the partial pressure of oxygen. The oxygen content in the atmosphere is 21% of volume giving $P_{O_2} = 0.21$ atm. At saturation in fresh water and at 5° C this corresponds to an oxygen concentration of 12.5 ppm. Henry's law gives

$0.21 = K \cdot 12.5 \qquad K = 0.0168$

$P_{O_2} = 0.0168\,(O_2 \text{ ppm})$

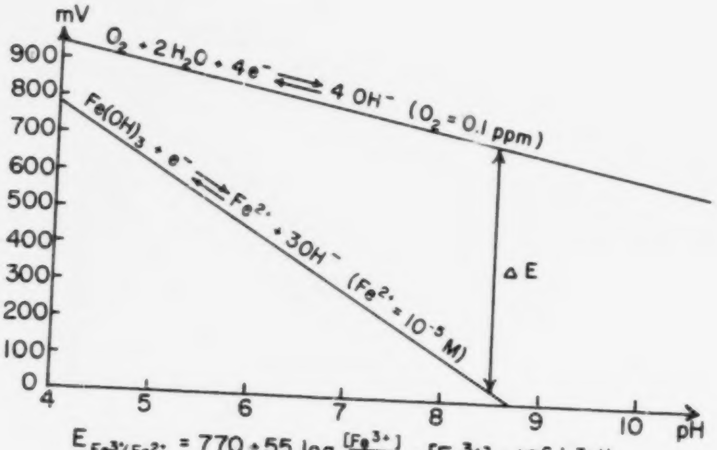
We introduce this expression into the above Nernstian expression and obtain

$$E = 376 + \frac{55}{2} \log \frac{(O_2 \text{ ppm})^{1/2}}{(OH^-)^2} \qquad \text{See Figure 5}$$

At a low oxygen concentration, e.g. 0.1 ppm, E_{O_2/OH^-} varies in the way indicated in Figure 5. From the values given in this table the following conclusions can be drawn:

a) Also, at as low an oxygen concentration as 0.1 ppm the resulting redox potential will cause ferrous iron to be oxidized to ferric iron.

b) The ferric hydroxide will be more stable at higher pH than at lower pH, as is seen from the difference between the two redox systems shown in Figure 5.



$$E_{Fe^{3+}/Fe^{2+}} = 770 + 55 \log \frac{[Fe^{3+}]}{[Fe^{2+}]} \; ; [Fe^{3+}] = 10^{6,1-3pH}$$

$$E_{O_2/OH^-} = 376 + \frac{55}{2} \log \frac{[O_2]^{1/2}}{[OH^-]^2} \; ; [O_2] \text{ in ppm}$$

FIGURE 5

CHEMICAL DIFFERENCES BETWEEN FERRIC AND FERROUS IRON

It might be of value to use the generalizations of Ahrland et al. (1958) who have outlined some fundamental differences between hard and soft metal acids. The hard metal acids tend to be in a high oxidative state and have their electron shells tightly bound to the atomic nucleus. The electrons are less able to polarize and the chemical bonds formed by these ions and their counterions show a high degree of ionic bonding. Typical hard acids are Mg^{2+}, Ca^{2+}, Al^{3+} and Fe^{3+}. The H^+ ion also shows the characteristics of a hard metal acid.

The soft metal acids are characterized by no oxidative state, or a low one. The electron shells are larger and more polarized and the chemical bonds also tend to be more polarized, leading to a high degree of covalency. Typically soft acids are Cu^+, Cd^{2+}, Hg^{2+} and Hg^{2+} . Fe^{2+}, Cu^{2+} and Zn^{2+} show characteristics that place them in between the hard and the soft acids. In their relations to anions, the hard and soft acids react differently. The following sequences of stabilities of complex ions are often found (Fig. 6). The atoms in the figure are assumed to be incorporated in the active sites of the anions.

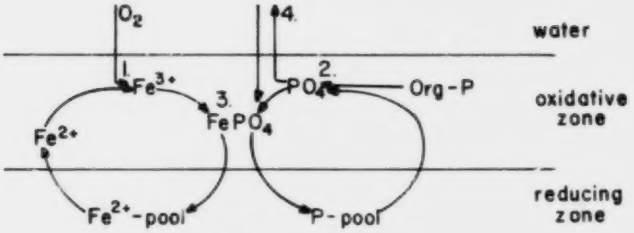
Hard Metal Acid: $N \gg P > As > Sb > Bi$
Soft Metal Acid: $N \ll P > As > Sb > Bi$

Hard Metal Acid: $O \gg S > Se > Te$
Soft Metal Acid: $O \ll S \sim Se \sim Te$

Hard Metal Acid: $F \gg Cl > Br > I$
Soft Metal Acid: $F < Cl < Br \ll I$

FIGURE 6 Frequent sequences of complex ion stabilities

We may now draw the following general conclusions concerning the iron processes taking place in a sediment system. Ferric ions will be the dominant iron species in those parts where excess oxygen is available. Since it is a hard acid, it will form stable bonds with OH^- and $O^-{-}PO_3^{3-}$. It may also be chelated by organic matters rich in carboxylic and amino groups. Deeper in the sediment layers, oxygen is absent in the interstitial water. Ferrous ions will dominate and, depending on the overall nature and composition of the sediment and the interstitial water, will appear in a free hydrated form or combined with available groups. Among these groups are the sulfide ions, the sulfhydryl group bound to organic material, $R{-}S^-$, and the carbonate ions of special importance. Diffusion will lead to the establishment of a kind of redox equilibrium during stagnation periods. The convection caused by bottom-dwelling organisms or by escaping methane will affect the rate of diffusion and thus the rates of the different reactions that are part of the equilibrium system. The processes are described schematically in Figure 7.



1. oxidation $Fe^{2+} + O_2 \longrightarrow Fe^{3+}$
2. organic-P $\xrightarrow{\text{bacteria}} PO_4$
3. $Fe^{3+} + PO_4 \longrightarrow FePO_4$ precipitate
4. nettodiffusion of PO_4

FIGURE 7

pH-DEPENDENCE OF FERROUS PRECIPITATES

Any tendency to form insoluble ferrous compounds during the establishment of anaerobic conditions in the sediment will cause the oxidation - from a kinetic point of view - to regress to ferric iron during instances when aerobic conditions are reestablished. The carbonate and sulfide are important in this connection. The solubility product of $FeCO_3$ is given by

$$(Fe^{2+})\ (CO_3^{2-}) = 10^{-10.7}$$

The carbonate is in equilibrium with hydrogen carbonate and carbonic acid. The solubility of $FeCO_3$ will thus be a function of pH as shown in Figure 7 (Weber, Stumm, 1963).

FeS reacts in the same way, the solubility of which is

$$(Fe^{2+})\ (S^{2-}) = 10^{-19.3}$$

The sulfide is in equilibrium with mono- and dihydrogen sulfide. The solubility of ferrous sulfide will vary with pH as is shown in Figure 8 (Hutchinson, 1957).

The above stresses the fact that the stabilities of all major ferrous compounds that are present in a sediment system will be functions of pH. It may be assumed that the combination of ferrous ions with any organic matter present in the sediment will follow the same general rule.

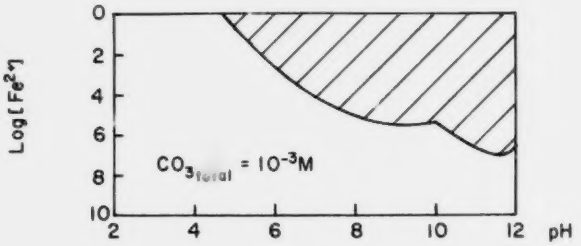
The following will take into consideration how the alkalinity of the total lake system will influence the equilibria outlined above.

ROLE OF HYDROGEN CARBONATE AND CARBONIC ACID

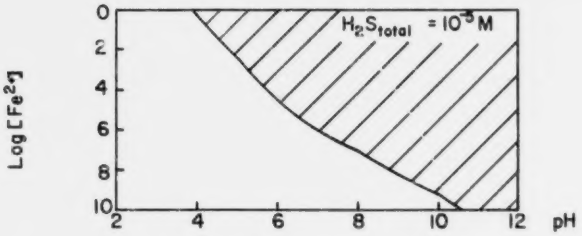
The pH of a lake system is influenced by a variety of factors. These include biochemical processes and exchange reactions between the atmosphere and the water and between the water and the sediment. The carbonate system will normally cover around 90% of the total buffer capacity. Organic matter of different origin will cover the remaining part. pH varies around pH 7 according to the relation

$$\mathrm{pH} = 6.52 + \log \frac{(\mathrm{HCO_3^-})}{(\mathrm{H_2CO_3})} \text{ at } 5^\circ \mathrm{C}$$

Discharge of sewage will increase not only the phosphorus concentration but in addition the concentration of bases. These bases are transformed into hydrogen carbonate by biochemical processes either in a treatment plant or in the receiving water. The increase in hydrogen carbonate concentration will raise the pH according to the above expression. A higher pH of the total mass will thus result. The change in alkalinity will influence both the trophogenic zone and the



Solubility of Fe^{2+} = f(pH) (10) in 10^{-3}M CO_3



Solubility of Fe^{2+} = f(pH) in 10^{-5}M H_2S

FIGURE 8

sediment system. The influence of pH on the redox potential of the O_2/OH^- and the Fe^{3+}/Fe^{2+} systems, upon the binding of phosphate to Fe^{3+} and to Al^{3+} and upon the solubility of various Fe^{2+}- precipitates has been mentioned previously. Moreover, all the trace elements will probably act in a way analogous to Fe^{2+} by being less soluble at higher pH.

The theory discussed in this paper illustrates that in order to enhance the capacity of the sediments to act as a sink for phosphorus, advantage could be taken of the fact that the binding of phosphate to ferric iron and to aluminum will increase in stability when the pH is lowered. The validity of this was tested previously for the total organic production in a series of experiments where the alkalinity of a nutrient-rich water was changed by the addition of controlled quantities of acid (Werner, 1969). The addition of acid-reduced hydrogen carbonate by partially replacing it with Cl^- or SO_4^{2-}. The results obtained clearly indicated that the primary production depends on the alkalinity and the pH, among other factors. Lowering of the alkalinity was accompanied by an approximately linear decrease of production, measured as the total amount of organic carbon. The following will describe supplemental experiments where the reactions of a sediment system to changes in the hydrogen carbonate concentration and the pH have been examined.

EXPERIMENTAL

Sediment and water were taken from the hypertrophic lake Lillsjön near Stockholm. The lake was heavily polluted by municipal sewage. The sediment was taken from the bottom by an Ekman dredge and the water was taken with a Ruttner sampler. The time of sampling, 6th of October, coincided with the fall turnover. 19% supersaturation, combined with a high pH, indicated that biological production was still taking place. Table I gives the analytical data of the sediment and the water sample.

50 ml of sediment and 450 ml of water were added to each of 24 - 500 ml Erlenmeyer flasks. The flasks were divided into six series and known amounts of 0.1000 M HCl were added to each group. The flasks were shaken slightly and kept at room temperature. Illumination was made so as to simulate the light conditions during fall circulation. The added amounts of HCl are shown in Table

TABLE I

Analysis of :	
Sediment	
suspended matter	50.3 g/l
total organic carbon	1.4 g/l
total iron	1.0 g/l
total phosphorus	0.108 g/l
Water	
dissolved oxygen	13.7 mg/l
temperature	8.8° C
pH	9.1
alkalinity (HCO_3)	1.58 mekv/l
total organic carbon	17 mg/l
total iron	0.120 mg/l
total phosphorus	0.500 mg/l
phosphate	0.235 mg/l

2. Visual inspection showed that the suspended matter settled much faster in the flasks with the largest amount of acid added: series 1 and 2 more than series 3 to 6. There was a difference seen within series 3 to 6, the matter in series 6 settling more slowly than in series 5, etc. This is what was expected. The added hydrogen ions neutralized excessive negative charges, thus making the repulsion between the particles less efficient. The particles agglomerated and settled more easily.

The aqueous phase was analyzed on day 11 (Table 2). Each value was the average of four replicates. It appears from this table that the alkalinity has decreased somewhat irregularly. This may be explained by the slow rate of certain reactions in which the solid phase takes part. Calcium and magnesium have partly been replaced by hydrogen ions, thus appearing in the aqueous phase. The iron is probably associated with colloidal or dissolved substances. Finally, the total phosphorus concentration has decreased in all the flasks.

TABLE 2

Analysis of Water After 11 Days

Series	Added Acid mekv/l	Alkalinity mekv/l	Fe µg/l	Total P µg/l	Ca mg/l	Mg mg/l
1	1.80	0	550	286	24.4	6.6
2	1.44	0	600	288	21.2	6.05
3	1.08	0.30	600	245	18.9	5.80
4	0.72	0.52	620	243	17.4	5.48
5	0.36	0.95	680	242	15.2	5.35
6	0	1.55	740	267	14.4	5.35

The flasks were placed in the dark at 20° C and covered with a membrane. This limited oxygen diffusion in the water, thus simulating the conditions in the near bottom water and the sediment during a period of stagnation. The elevated temperature was selected in order to speed up certain reactions. Oxygen and pH were analyzed after a total of seven days in the dark (Table 3).

The flasks were analyzed again after an additional period of 31 days in the dark under the same conditions (Table 4).

The iron concentration had dropped considerably in all instances, and could be detected only in the series having the alkalinity reduced to zero. In the series with a measurable alkalinity, the alkalinity had decreased in all instances. One of the reactions that probably had taken place in the sediment, causing the alkalinity to decrease, is the following:

$$\mathrm{FeS} + 9/4\ \mathrm{O_2} + 2\ \mathrm{OH^-} + 1/2\ \mathrm{H_2O} \rightarrow \mathrm{Fe\ (OH)_3} + \mathrm{SO_4^{2-}}$$

The pH values in the extreme series 1 may very well be considered only of interest as illustrations of the discussed theory, as may be those of series 2 and 3. The concentration of hydrogen carbonate is zero in these series. The values for oxygen indicate that bacterial decomposition of

TABLE 3

Analysis of Water After 7 Days In The Dark

Series	pH	O_2 mg/l
1	4.50	4.1
2	5.31	4.2
3	5.45	3.3
4	6.45	4.5
5	6.83	4.0
6	7.15	4.8

the organics continues in all series. A consequence is that carbonic acid and carbon dioxide are present in excess. In accordance with theory, the phosphorus concentration shows the largest drop in the series having no alkalinity and a low pH. Also the series having a measurable alkalinity shows an apparent relationship between the parameter and the phosphorus concentration. The lower the pH of the water the less phosphorus will be in solution (Fig. 9), thus indicating that phosphorus will be absorbed by the sediment system in a direct dependance upon pH when aerobic conditions are maintained.

It is felt that the above discussion and the results reported may assist in the interpretation of the phenomena associated with the discharge of sewage to natural waters of different types concerning alkalinity and character of sediment system.

TABLE 4

Analysis of Water After 40 Days In The Dark

Series	Alkalinity mekv/l	Total Fe µg/l	P µg/l	PO_4-P µg/l	pH	O_2 mg/l
1	0	170	10	10	4.1	2.35
2	0	60	16.1	13.2	4.8	1.73
3	0	30	24.4	18.0	5.3	2.02
4	0.32	trace	79	61	6.1	1.73
5	0.85	trace	92	84	6.6	1.50
6	1.44	trace	112	108	7.0	1.60

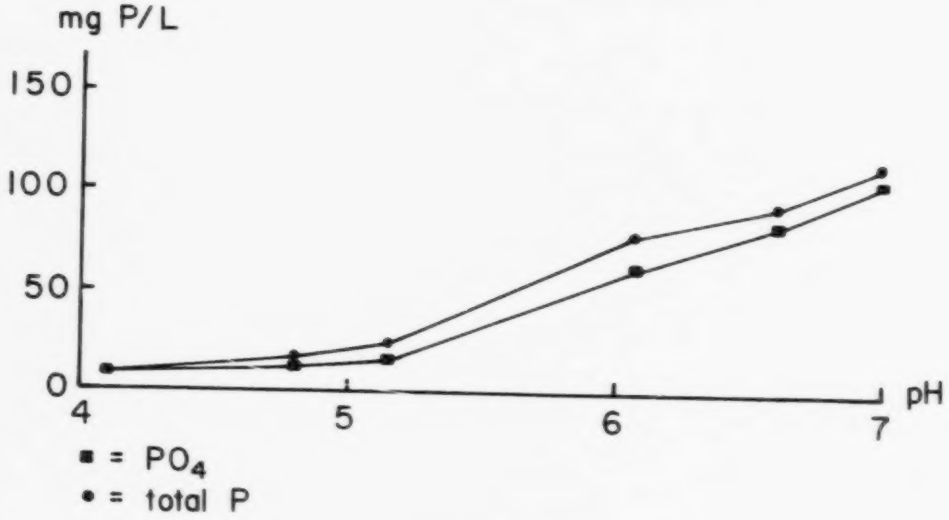


FIGURE 9

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CRITICAL REVIEW OF PAPERS ON RECEIVING WATERS

Peter A. Krenkel

INTRODUCTION

Temperature has profound effects on biological, chemical, and physical processes. It is particularly appropriate to investigate these effects under cold weather conditions, inasmuch as a paucity of information exists regarding waste treatment and water quality management in the Arctic.

The organizers of this conference are to be commended in gathering together such an eminent group of scientists and engineers, and it is a great honor to have the opportunity of discussing a portion of the papers presented.

I have attempted to be as objective as possible in the following discussions, and it is hoped that the authors involved will accept my comments as constructive discussion rather than criticism.

Synoptic Study of Accelerated Eutrophication in Lake Tahoe, California-Nevada

Charles Goldman

Lake Tahoe is probably one of the most beautiful lakes in the world and changes as described by Dr. Goldman are quite obvious to me, since I spent many days there some 20 years ago. The real problem is, of course, people.

Measurements included primary productivity of phytoplankton and periphyton, species composition, biomass, and biotic diversity. I believe the productivity information is of particular significance inasmuch as the methodology utilized appears to yield maximum information in a minimum of time and effort.

The results of the study demonstrate the effects of the added nutrients on the receiving water and certainly demonstrate the need for control measures.

The author is to be complimented on the analysis of his data. There is little doubt that the future of water pollution control will dictate more quantitative work from the biologists, as is the case with Dr. Goldman's work. The use of statistics and parameters, such as a species-diversity index, are to be encouraged.

The South Basin of Lake Winnipeg - An Assessment of Pollution

Jo-Anne M. E. Crowe, Canada

The author has presented data in an attempt to demonstrate that Lake Winnipeg is suffering from an increase in the rate of eutrophication. Three indices were utilized in this endeavor; physical and chemical measurements, benthic organisms, and fish.

There is little doubt that the lake, as described by the author, is suffering from the effects of pollution; however, the data presented is not conclusive with respect to changes noted in severely affected eutrophic lakes. It is unfortunate that no measurements of phosphorous or nitrogen were available that could be assessed for their role in the apparent changes occurring in the lake.

The author mentions the severe oxygen depletion noted in the Lake Erie basin and the increases in various ionic species indicative of the conditions in that basin. She then states that dissolved oxygen values in the lake are usually above 80% of saturation and that the only ionic component showing an increase was calcium. The Secchi disc readings were significantly reduced in the 39-year period, however.

Examination of the data fails to reveal significant differences in the chemical measurements at first glance, even of calcium. It would be interesting to subject the data to a statistical analysis in order to determine the statistical significance of the changes, if any, on a quantitative basis. Mere comparison of averages means little with this type of data. Differences in sampling techniques should also be accounted for.

Measurements of the phytoplankton would have been interesting, as would discussion and analysis of some of the chemical data.

The only really significant changes that were noted were an increase in benthic density, a shift to more tolerant forms of bottom organisms, and a decrease in the fish catch. Again, a statistical analysis of these factors would be of considerable value in determining the real significance of these changes.

Another informative evaluation of the data would be to utilize some form of species diversity index with respect to the bottom organisms, which will be mentioned later.

While the fishing yield has been significantly reduced, the author states that this reduction could be attributed to "over-fishing" and the use of illegal methods.

An additional point of interest is the possibility of mercury playing a role in the change in the benthos. The toxicity of mercury is extreme, and if the concentrations in the fish have "exceeded 0.5 ppm", one cannot help but wonder what the concentrations might be in the organisms being ingested by the fish.

Eutrophication in Some Lakes and Coastal Areas in Finland With Special Reference to Polyhumic Lakes

Pasi O. Lehmusluoto, Finland

A good review of the phytoplankton production in Finland and in arctic areas was presented by Dr. Lehmusluoto. He points out that in Alaska, almost half of the annual phytoplankton primary production may occur beneath the ice sheet in the spring time. He suggests that the mean maximum primary production rate per unit volume in the growing season measured *in situ*, or in constant light, should be used as an index for the many kinds of water bodies to give a relative, but objective, index for the phototrophic level of the water. It is also demonstrated that the direct comparison of trophic states of humic and nonhumic lakes is impossible, as the classification of polyhumic lakes on the basis of phytoplankton production cannot give an objective result.

Polyhumic water in Lake Hakojarvi showed that nitrogen was the primary limiting nutrient and was also a primary limiting nutrient for bacterial growth. The addition of nitrogen and phosphorus caused a large algal growth. Seven *in situ* plastic test cells were used, the cells being ten meters deep, 1.2 meters in diameter, and having open tops with a water volume of 12 m^3. Nutrient additions ranged from 0.05 to 1.0 mg/l nitrogen and 0.005 to 0.1 mg/l of phosphorus. Phytoplankton primary production was measured by the carbon 14 method. Phosphorus alone did not cause significant eutrophication and there were not any large changes when only nitrogen was added. In the cells where both nitrogen and phosphorus were added, the greater the addition of nutrients, the larger the increase in the phytoplankton production.

Humus may be important in the total production of polyhumic waters due to light extinction. It is also proposed that humus, as such, or transformed to bacterial biomass, may serve as food for zooplankton and, thus lead to relatively high fish production. It should be noted that bacterial numbers in humic waters are normally comparable to numbers in eutrophic lakes.

In Lake Saimaa, phytoplankton production has been studied. The lake receives 1,200 m^3/day of biologically purified domestic sewage and 260,000 m^3/day of mechanically purified pulp mill waste, which is 44% sulfite and 56% sulphate liquor. The domestic sewage stimulated the algae growth. Pulp mill effluents were almost lethal to algae at 10% effluent concentrations, and in 1% and 0.1% effluent concentrations, the algae growth was slightly stimulated. Pulp mill effluents seemed at first to hinder phytoplankton primary production, but caused pronounced eutrophication about 9 km from the outfall. Both domestic sewage and pulp mill effluents do cause eutrophication. The maximum eutrophication caused by the pulp mill effluents is more intensive than that caused by the domestic sewage, though it occurred far from the discharge area.

Eutrophication is a problem in the coastal area in Finland, as well as in lakes. The major problems are near the cities on the coast and are important recreational areas. Blue-green algae blooms occurred during late summer and restricted the use of the waters.

The phytoplankton production 1 km from the outfall was about 30 times higher than that at 12 km distance from the outfall. Annual primary production near the outfall was 5 times higher than in the unpolluted areas.

In conclusion, it is important to consider all the possible efforts to reduce nutrient input into receiving waters, in any form, in order to avoid overeutrophication, of waters.

It would be interesting to hear the speaker's comments on the recent hypothesis of Knentzel, which has caused considerable concern in the U. S.

Knentzel (1969) concluded that phosphorus and nitrogen were not the limiting factors in eutrophication, but instead, organic material and the resulting production of CO_2 by bacteria. Knentzel's conclusions were based on the following observations:

1. In natural water, blue-green algae and bacteria are always found in close association. Separation is detrimental to algae.
2. Massive algal looms are always associated with excessive amounts of decomposable organic matter.
3. CO_2 is the major nutrient for algal growth. 2 gms CO_2 are required for one gram algae.
4. Large amounts of CO_2 required cannot come from the atmosphere or land-dissolved carbonate salts.
5. Bacteria can supply 20 mg/l CO_2.
6. P is widespread in nature and algal blooms have been documented where $P < 0.01$ mg/l; and no blooms have been documented where $P > 0.01$ mg/l but no organics.

It is interesting to note that recent work by the Federal Water Quality Administration Laboratory in Athens, Georgia, appears to lend credence to Knentzel's theory (Kerr, et al., 1970).

Observations on the Recovery Process in a Lake which had Earlier Received Waste from an Ore-Dressing Plant

Bengt Ahling, Sweden

The objective of this paper was to compare the physico-chemical and biological parameters of Lake Balsjon in Central Sweden before and after the shutdown of an ore dressing plant. Initially, the parameters from Lake Balsjon were compared to a control lake before shutdown. The question asked was "Can a recipient of some waste satisfactorily recover, purify itself to the point at which its water can be used for more desirable purposes, within a reasonable time?"

The ore dressing plant's principal ores were hematite and magnetite. The process involved gravity and wet-magnetic concentrations, hence large amounts of iron were deposited in the lake.

Table 1 shows a comparison of the physico-chemical parameters presented in this paper.

The most interesting part of the paper, and possibly the most important factor, is the development and changing of the ecosystem within the lake. Originally the lake was a 'sterile milieu'. Initially, there was a noted absence of sessile living lake-bed organisms and submerged plants which was due to the reduced penetration of sunlight. No zooplankton were found whatsoever.

TABLE 1

Parameter	Before Control Lake	 Lake Bal	After Control Lake	 Lake Bal	Comments
1. Turbidity	<100 ZP units	>1000 ZP units	100 ZP units	400 ZP units	
2. Colour	30 mg Pt/l	650 mg Pt/l	30 mg Pt/l	30 mg Pt/l	Returned to normal hue
3. Secchi disk transparency	2-4 meters	Few centimeters	3 meters	2 meters	Due to reduced turbidity
4. pH		7.8 - 8.1*		Reduction in pH (spring) Increase in pH (fall)	Primary production Increased which increases pH
5. Conductivity		7 times the control lake		3 times the control lake	Actual values undecipherable
6. Total iron, mg Fe/l	<.5	1-10	<.5	.5	
7. Oxygen		90% sat		Still high	Bottom sample showed oxygen deficit down to 17%
8. Consumption of permanganate mg KM_nO_4/l	20	10	30	30	Increase showed increase in organic material in lake
9. Nitrogen and phosphorus					No change No values reported in paper

* After dilution with Lake Balsjon waters

After the plant shutdown, consequently reducing the turbidity of the water, the lake was again suitable for biological growth. The occurrence of phytoplantkon was markedly increased, especially diatoms. Organisms that were completely absent initially began to appear in abundance, e.g. chrysomonad rhodomonas and dinobryon divergens. Besides increasing the number of individuals in a species, the number of different species increased. Two years after the plant shutdown the composition of zooplankton was almost identical to that of the control lake. Also, it was noted that fauna normally living on the lake bed was in the process of being built back up.

After observation of the physico-chemical parameters, one is led to believe that Lake Balsjon is returning to its normal state as compared to the control lake. However, the values reported in the paper are very general in most respects. Yet, the building up of a balanced biosystem is quite evident by the changing biological picture. Not only is the number of individual species increasing but the number of species is also increasing. Therefore, one can conclude for this particular lake that the physico-chemical aspects are returning to a normal pattern slowly, while the ecosystem of the lake is returning to a balanced system much faster.

As previously mentioned, the use of some form of a "Species Diversity Index" for comparative purposes would add considerably to the conclusions that could be made from this study.

Depletion of Oxygen by Microorganisms in Alaskan Rivers at Low Temperatures

Ronald C. Gordon, USA

This paper demonstrated that bacterial degradation occurred in subarctic rivers at temperatures of 0° C to 20° C. As the incubation temperature was decreased, a lag phase increased, but extensive metabolic activity was observed at all temperatures.

The rate also decreased with decreasing temperatures as might be expected. The rate of activity was affected by the nature of the substrate, primary effluent, and rapid D.O. depletion was shown at all temperatures, while secondary effluent showed activity at 10° C and 20° C but none at 0° C, probably indicating the effect of nitrification, i.e., the high temperature dependency for nitrification yielded no growth at 0° C.

Psycrophillic bacteria grow well at low temperatures, but the rate of activity increases with increasing temperature.

It is obvious that the 5-day 20° C BOD test is not a good measure for pollution control at low temperatures because of possible different mechanisms of metabolism. It would appear that the "yardstick" for pollution in cold regions should be some form of total carbon analysis or at least that the BOD values measured should be at the temperature of the receiving water.

One wonders why the author did not extend the data analysis to include determination of the effects of temperature on the rate constants and a comparison of the rate constants themselves. It would appear that the data required for this information has already been acquired and all that is needed is some mathematical analysis and then a quantitative presentation of the results.

This information would be quite useful inasmuch as the characteristics of the oxidation rate constants at low temperatures need elucidation.

Prediction of Dissolved Oxygen Levels in the South Saskatchewan River in Winter

Robert C. Landine, Canada

This paper is interesting inasmuch as some of the differences in stream analysis in cold climates over warm weather conditions are discussed. The low temperatures, ice cover, lack of local inflow, high oxygen saturation values and low reaction rates are all different from those usually encountered.

The approach described is more or less a standard one, with modifications for cold weather treatment. It should first be pointed out that utilization of this type model requires steady state conditions, an assumption that is rendered invalid because of the hydroelectric power plant operations.

In addition, the goodness of fit of the proposed model is questionable inasmuch as 4 sampling points are hardly sufficient for a 202 mile reach. Also, since the change in Dissolved Oxygen was less than 1 mg/l in ~12, almost any coefficients would produce a fairly good fit with the relatively low organic loadings used. Thus, there is no assurance that the model will accurately predict effects of major changes in flow and/or loading conditions.

The apparent anomaly wherein the assumed values of K_1 had a negligible effect on the deoxygenation rate in the river was partially due to the incorrect assumption that the bottle K_1 was identical to the river deoxygenation coefficient, K_r. This is very seldom true inasmuch as oxidation in a bottle is hardly similar to that occurring in a river.

The ultimate first-stage BOD is a fixed parameter and should be determined independently by a long term BOD test. It can be estimated from short term BOD's if the bottle coefficient, K_1, is also determined. L is not a function of K_1 as implied by the author. Once L is determined, the rate of deoxygenation in the river is assumed to be proportional to L and K_d, which should be determined in the river.

It should also be noted that the author's use of the English reaeration equation will probably not be adequate for the conditions of the Saskatchewan River. This equation was empirically derived from very small streams and brooks and has been shown many times to be inapplicable to rivers the size of the Saskatchewan. If the author had to use an equation of this type, he would have better success (optimistically) using Churchill's equation (1962):

$$k_2 = 5.026\, U^{-0.969}\, h^{-1.67}$$

which was derived from what is considered to be the best stream data ever taken. In either case, the use of empirical equations should be subjected to extreme cases in order to avoid gross errors.

The author might well have applied a more rational equation such as the one progressed by Thackston & Krenkel (1969):

$$k_2 = 0.000125\,(1+F^{1/2})\,\frac{\bar{U}_*}{h}$$

$$F = \frac{\bar{U}}{gh} \qquad \bar{U}_* = hSeg$$

which has a theoretical base and has been successfully used in field measurement.

The comments herein should not detract from the effort put forth by the author inasmuch as obvious limitations were imposed because of a lack of adequate time and funds. As is the case in many water quality investigations, the results must be in accord with economic limitations of data collection.

Pollution - A Biological Study of Some Receiving Waters in Hokkaido

Matsunae Tsuda, Japan

The objective of this paper was to study the pollution effects on the biota of the Ishikari River. The paper gives a fairly good descriptive and qualitative piece of work but gives no real definitive idea of the total effect on the biota, the degree of damage, or any way of comparing this data with other work.

A major weakness in the paper is that the writer has but one source to compare and give background to the study. A major aspect appears to be concerned with population dynamics. It should be noted that in 1964, Silvey and Roach published a definitive paper on population dynamics in fresh water systems using ten years of accumulated data. There has been a great deal of work dealing with species diversity as related to pollution perturbances. Two fine papers were published by Odum, et al. (1963), and Pearson (1967), and demonstrate the variances of species diversity attributed to various pollutants.

The author did mention how and with what equipment the data was taken. Since no reference was made to sampling techniques and methods of analysis, scientifically, this data has little meaning either qualitatively or quantitatively. It is obvious that sampling and analysis techniques should be described so that the degree of accuracy and precision of observed data may be ascertained.

If the microbenthotic organisms were analyzed by a microscope, the species diversity could easily be obtained. This would yield a somewhat quantitative answer rather than the qualitative answer, "There were numerous organisms or just a few species were observed". A common species diversity formula used is.

$$I = -\Sigma_i\, Pi\ Log_e\, Pi$$

where $Pi = n_i/n$

n_i = number of individuals per species

N = total number of species

When working with microorganisms such as zooplankton, etc., Copeland (1966) used a simpler method to obtain a species diversity, which is:

$$I = \text{number species}/1000 \text{ microorganisms}$$

It is also of value to determine some abiotic analyses such as TOC, BOD, COD, and dissolved oxygen, in order to obtain a better picture of the water quality. This additional information would allow interpretation of the biological data with greater significance than only qualitatively.

Chemical Effects of Decomposing Salmon Carcasses on Aquatic Ecosystems

David Brickell & John Goering, U.S.A.

The authors have presented some interesting data in an attempt to explain a most complex system. The toxic nature of ammonium ions in the water is certainly of concern; however, the authors did not measure pH, which has a rather drastic effect on this toxicity. It is possible that a high concentration of ammonium ions at a low pH value will be amenable to fish life, but if the pH value is increased the toxicity will probably increase. It has been reported that the toxicity of a specified concentration of ammonium compound tested with fishes increased by at least 200% between pH 7.4 and 8 (1937). It should also be noted that the toxicity of ammonia is reduced in the presence of CO_2 and increased with low concentrations of oxygen.

The authors did not mention the stratification occurring in some of the sampling areas. Examination of the temperature and the salinity data demonstrates density differentials sufficient to maintain a rather stable stratified flow regime. This would help to explain the observed increases in nitrogen and decreases in oxygen that occurred at these lower levels. With the new emphasis on water pollution control, the nitrogen problem will become more significant as increased treatment introduces effluents of a more stabilized carbonaceous water and readily available organisms that can oxidize ammonium to nitric and nitrite to nitrate. It would therefore be quite useful for the authors to attempt to quantitize the nitrogen phenomena that is the subject of their paper.

Since it is an autocatalytic reaction, nitrogen transformation can be described by:

$$\frac{dc}{dt} = KC(N\text{-}c)$$

where: C = concentration of ammonium or nitrate
N = concentration originally present
K = reaction coefficient

Wezernak and Gannon (1968) have utilized this model with success on several receiving waters. Knowing the time of passage, samples are collected at a minimum of three points in the study section. One sample is collected at the beginning of the study section to determine the upper limits of ammonium and nitrite oxidation and to determine the amount of inorganic nitrogen oxidation between the starting point and the other two stations. Of the remaining two samples, one is taken near the beginning of the study section and one near the end.

The effect of temperature on the process is evidently quite pronounced inasmuch as lower temperatures appear to significantly decrease the rates of nitrification. It would be interesting to observe the rate constants for the nitrification processes under the cold conditions of this study. Optimum temperatures for the nitrifying bacteria have been reported in the range of 28° - 36° C, considerably above the temperature of this study.

Self-Purification of Polluted Lakes in Temperate Regions - Phosphorus Binding Mechanisms

Jan Werner, Sweden

The author discussed a mechanism for phosphorous removal in lakes. By using solubility products and thermodynamic relationships, he concluded that ferric ion will be the dominate species of iron in oxygen-containing waters, and ferrous ion will dominate in oxygen-deficient regions. He also discussed the pH dependence of ferrous precipitates. It was concluded that phosphorous removal could be improved by a lowering of the pH to 4 because of the increase in stability of the binding of phosphate to iron. These conclusions can also be found in work by Morgan and Stumm (1964), Hem and Cropper (1959), Anon. (1970), and others.

It should be noted that a substantial fraction of iron in lake waters is present in suspended form and the insoluble ferric ion sediments into the hypolimnetic waters. The rate of sedimentation is influenced by many factors, probably the most significant being the colloidal chemical nature of the ferric precipitate.

The ferrous ion may be soluble up to a few mg/l, depending on alkalinity and pH and is of significance in the oxygen-deficient hypolimnion. The ferric ion not reached in the overlying waters will be reduced at the mud-water interface.

The progressive accumulation of phosphate in the hypolimnion may be partly attributed to plankton, but other factors are involved. There is usually more iron than phosphorous in lake water, leading to the formation of ferric-hydroxy-phosphate in the upper waters immediately after circulation. Hutchinson (1947) has reported that an oxidized mud surface not only holds phosphate but prevents diffusion of phosphate from deeper mud layers, as ferrous iron is always in excess and when oxidized, precipitates the phosphate.

The author did not mention the amphoteric properties of ferric hydroxide, which may play a role in the process: Isoelectric point at a $pH = 5.5$, $pH > 5.5$ the species is more negative, $pH < 5.5$ the species is more positive. Also, the various exchange mechanisms in the sediment and on the hydrous metal oxides should be investigated. Since silicates interact with iron in a manner similar to phosphates, their role should be elucidated.

The author also failed to take into account the species change of the inorganic phosphate with changes in pH. At very low pH's, H_3PO_4 is the major species, at approximately a pH of 4, $H_2PO_4^-$ is the major species, and so on until at pH's of 11 to 13, orthoohosphate becomes the predominate inorganic phosphate species. At a pH of 4, the ferric cation will not precipitate as $FePO_4$ but as $Fe(H_2PO_4)_3$, and this precipitate will have a completely different precipitation solubility product than ferric phosphate. In a chemical sense, we are dealing with a dynamic and unsteady system, and, therefore, the rates of precipitation equilibrium studies carried out in this paper.

Finally, the influence of biological activities on the chemical reactions described should be investigated. There is little doubt that the biota plays a significant role in determining the distribution of chemical constituents in a lake.

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THE INFLUENCE OF TEMPERATURE ON THE REACTIONS OF THE ACTIVATED SLUDGE PROCESS

Pál Benedek and Péter Farkas

The rate of biochemical reactions is increased by an increase in temperature and vice versa. However, while the microbial processes of fermentation industries are carried out at constant temperature, biological waste treatment is exposed to temperature changes predetermined by the prevailing climate. This fact, of course, exerts an influence on purification efficiency. Namely, detention times in the waste treatment facilities are fluctuating in summer and winter between the same limits, while the rate of the reaction - and so the purification efficiency - decreases considerably with falling temperature.

Temperature dependence of the biological treatment process can be described with a simple mathematical model only between certain limits because it is a complex phenomenon. It is well known that various technological modifications of the activated sludge process react diversely to changes in temperature, the aerated lagoons being most sensitive to the cold (Table 1). Explanation of this phenomenon was given by Eckenfelder (1966) who supposed that the rising temperature results in anaerobic conditions in the inner part of the flocs (oxidation lagoons have dispersed bacterial systems, whereas activated sludge plants have flocculated ones). Thus, reaction rate increase with temperature is compensated for in a negative sense by the smaller bacterial mass taking part in aerobic substrate removal processes.

The situation is further obscured by observations of Wuhrmann (1964), indicating that not only the rate of the reaction, but also its stoichiometric constants (g O_2 consumed/g substrate removed) are changing with temperature. Downing (1968) pointed out that this was caused predominantly by the intensive temperature dependence of nitrification. (At temperatures above 10° C nitrogen metabolism produces NO_3^-, whereas at lower temperatures NH_4^+ is formed, the latter process needing less oxygen.) It cannot be excluded, however, that the fraction of substrate, being oxidized for energy yield, is also changing with the temperature.

The aims of this study are to reveal the physico-chemical basis of temperature dependence measurements, and based on the kinetic interpretation, to measure the temperature dependence of individual substrate removal and oxidation processes.

KINETIC INTERPRETATION

In a non-steady state system, the reaction rate, $\frac{dS}{dt}$, is a function of temperature, T° (in degrees Kelvin), and reactant concentration, S_t, at time t, as described below:

where $\frac{dS}{dt}$ is $f(T^\circ, S_t)$ and K is $f(T^\circ)$;

$$\frac{dS}{dt} = K S_t^n \qquad (1)$$

where K is the reaction constant (being a function of temperature) and n is the order of reaction.

TABLE 1

Temperature Dependence of Biological Waste Treatment Processes

Process	Temperature range, C°	Q_{10}	θ	Source
Activated sludge	0 - 20	1.00 - 1.48	1.000 - 1.040	Eckenfelder, 1966
Trickling filter	0 - 20	1.41	1.035	"
Aerated stabilization pond	0 - 20	1.96 - 2.15	1.07 - 1.08	"
Paper and pulp mill wastes	2 - 30	1.35 - 1.56	1.031 - 1.046	Carpenter et al., 1968

To describe temperature dependence of the rate of enzyme-catalyzed biochemical reactions - where one cannot speak of an "order of reaction" in the strict physico-chemical sense - Equation 1 modifies to:

$$\frac{dS}{dt} = A = K \cdot \chi \cdot f(S_t) \tag{2}$$

where both $\frac{dS}{dt}$ and A are $f(T^\circ, S_t)$ and K is $f(T^\circ)$;

where χ is the concentration of biocatalyst (enzyme or active bacteria) and $f(S_t)$ refers to the substrate dependence of reaction rate, that is, "biological activity", A.

Based on the foregoing generalized relationships, the Michaelis-Menten model (Pearson, 1968; Benedek and Farkas, 1968) can be written:

$$A = V_s^{max} \cdot \chi \cdot \frac{S_t}{S_t + K_m} \tag{3}$$

where $K = V_s^{max}$ as $f(T^\circ)$;

$$\text{so } K = V_s^{max} = (\text{constant})\, e^{\frac{\Delta H}{RT}} \tag{4}$$

and

$$f(S_t) = \frac{S_t}{S_t + K_m} \tag{5}$$

Equation 4 represents the Arrhenius equation, where ΔH is the "activation energy" expressed in calories (Thimann, 1964; Johnson, et al., 1957; Ingraham, 1962). V_s^{max} is the maximal value of specific substrate removal rate defined as $\frac{1}{\chi} \cdot \frac{dS}{dt}$. Equation 5 is the so called "enzyme kinetic function", where K_m is the Michaelis constant.

Supposing that $S_t < K_m$ (which is the case under normal operating conditions in the activated sludge process), the generally known first-order reaction rate can be attained:

$$A = \frac{V_s^{max}}{K_m} \cdot \chi \cdot S_t = K' \cdot \chi \cdot S_t \tag{6}$$

where K' is a first-order reaction constant and $n = 1$.

From the point of view of temperature dependence measurement, it is important that χ and $f(S_t)$ should be constant. In this case, from Equation 2 it follows that

$$\frac{A_1}{A_2} = \frac{K_1}{K_2} \tag{7}$$

where the subscripts 1 and 2 refer to A and respective K values measured at two different temperatures. Metabolic rate measurements must be made at different temperatures with the substrate being removed on the course of a zero order reaction - in this case, reaction rate does not depend on substrate concentration, or S_t should be the same constant value for all measurements performed. Of course, sludge samples have to be taken from the same system, thus assuring constant bacterial concentration (χ), too. Adopting the above test conditions, the actual biological activity (A), can be measured instead of the respective K values, which are difficult to develop. Metabolic reaction rate parameters like substrate and endogenous respiration, substrate removal, dehydrogenation, etc., are regarded as different forms of A.

To calculate ΔH from Equation 4 and 7, it follows:

$$\Delta H = \log \frac{A_2}{A_1} \cdot 2.3\ R \cdot \frac{T_2 \cdot T_1}{T_2 - T_1} \tag{8}$$

As an index for temperature dependence, the coefficient θ is also used with the following definition of Eckenfelder (1966):

$$\frac{A_2}{A_1} = \theta^{(T_2 - T_1)} \tag{9}$$

A further symbol characterizing temperature dependence is Q_{10}, the factor indicating how many times the reaction will increase if the temperature is raised by 10° C (Thimann, 1964).

$$Q_{10} = \frac{A_{(T^\circ + 10)}}{A_{T^\circ}} \tag{10}$$

An interrelation between the three parameters is given below, and deducted in the Appendix:

$$\log Q_{10} = 10 \log \theta = \Delta H\ (2.54)\ (10^{-5}) \tag{11}$$

For steady state systems, like continuously working activated sludge plants, hydraulic residence time V/q must be employed instead of physical time, t, and S_e, effluent substrate concentration, instead of S_t.

The term describing removal activity follows from the materials balance, Equation 2:

$$A = \frac{q}{V}\ (S_o - S_e) = K \cdot \chi \cdot f\ (S_e) \tag{12}$$

where V is the aeration compartment volume, q is the waste flow, and S_o is the substrate concentration in the raw waste. The same rationale is employed as in Equations 2-6, observing the above indicated changes in t and S_1, respectively. Assuming a first order reaction analogous to Equation 6, the expression can be written as:

$$\frac{q}{V}(S_o - S_e) = K' \cdot \chi \cdot S_e \tag{13}$$

where $K' = f(T^\circ)$,

from which the value of K' can be calculated graphically after rearrangement:

$$\frac{S_o}{S_e} = 1 + K' \cdot \chi \frac{V}{q} \tag{14}$$

and plotting the respective K' values versus T°, using either the rearranged Equation 8 or the logarithmical form of Equation 9, the temperature dependence may be calculated (Carpenter et al., 1968).

Strictly speaking, in Equations 12-14, the term $S_e - S_e^{min}$, representing available substrate concentration, should be taken for S_e, where S_e^{min} is the so-called "residual BOD" remaining in the effluent after prolonged biological treatment (Benedek and Farkas, 1968 and Benedek et al., 1968). If the "residual BOD" is neglected, S_e will exert less change in the temperature function than the "available food concentration", the latter having the proper direct relationship for removal rate.

EXPERIMENTAL

Measurements were performed on endogenous sludge that had been aerated for some days previously. Sludge samples were taken from plants treating municipal and pharmaceutical wastes, respectively. Some tests were also made with sludges grown on pig farm wastes or phenolic wastes. The temperature range was 0° - 25° C. For a set of measurements, one-liter sludge samples were taken from the same system, chilled to the desired temperature and thermostatized during the respiration or activity (substrate removal) measurements. All measurements were done in non-steady state (batch fed) systems. Temperature dependence of the following processes were examined:

Colloid removal rate,

Oxidation rate of adsorbed colloids,

Oxidation rate (substrate respiration) of dissolved substrate,

Endogeneous oxidation (respiration) rate.

In the case of dissolved substrate, the amount of O_2 consumed per gram of substrate added was also determined using the short term oxygen demand (STOD) process of Vernimmen et al. (1967).

Colloid removal rate was measured mixing a stock colloid substrate solution (caseine, soap or starch) with endogenous sludge of about 3 g/l MLSS. It was settled for about 5 minutes, then the supernatant was filtered and S_t measured as COD. The S_0 concentration for $t = 0$ was calculated from the well-known mixing formula.

Removal activity, A, was calculated as follows:

$$A = \frac{\Delta S}{t} = \frac{S_0 - S_t}{t} \tag{15}$$

The further fate of adsorbed substrate was followed by respirometry. Because the biooxidation rate of starch was found to be very low (substrate respiration could be neglected in comparison to endogenous respiration), this test was carried out using a washed caseine suspension. The mixed liquor containing the caseine was aerated for an hour after mixing to allow the dissolved fractions to oxidize, which was indicated by an intensive drop in respiration. Respiration rate measured after the removal of the dissolved fraction was regarded as the net sum of the substrate respiration due to the biological oxidation of caseine and the endogenous respiration.

Substrate removal rate was measured by means of the Farkas (1968) activity measurement procedure. Most substrates (acetate, sucrose) were removed by a zero order reaction, thus assuring $f(S_t)$ to be constant. Substrate load was kept constant for each set of measurements.

Respiration measurements were done with membrane-coated D.O. probes, measuring first the endogenous respiration at the desired temperature, then adding a constant amount of the substrate to be examined and measuring the total respiration. Substrate respiration was calculated as the difference between total and endogenous respiration.

The log reaction rates (A) were plotted against $1/T^\circ$, (see Equation 6), where reactions obeying the Arrhenius law give straight lines. In all cases where this did not occur, the Q_{10} and θ values were calculated for the 0° C - 10° C and 10° C - 20° C temperature ranges, using Equations 10 and 11.

RESULTS

The rate of starch removal - which was a purely adsorptive process - did not show any significant thermal dependence. From Figure 1 the ΔH value was calculated and found to be 364 calories, which is much less than ΔH values encountered in biochemical substrate removal processes.

The plot of the biochemical oxidation of adsorbed caseine suspension (Figure 2) indicates that biooxidation of caseine obeys the Arrhenius law.

The metabolic rate of dissolved substrates did not give straight lines in the Arrhenius system, as can be seen in Figure 3 in the case of activated sludge grown on municipal waste and fed acetate. Nearly the same result was obtained with the same sludge and sucrose (Fig. 4) and activated sludge grown on a pharmaceutical waste, fed phenol and acetate, respectively (Fig. 5).

In contrast to the above, endogenous oxidation always followed the Arrhenius law. The thermal dependence of endogenous respiration of four different kinds of activated sludge is plotted in

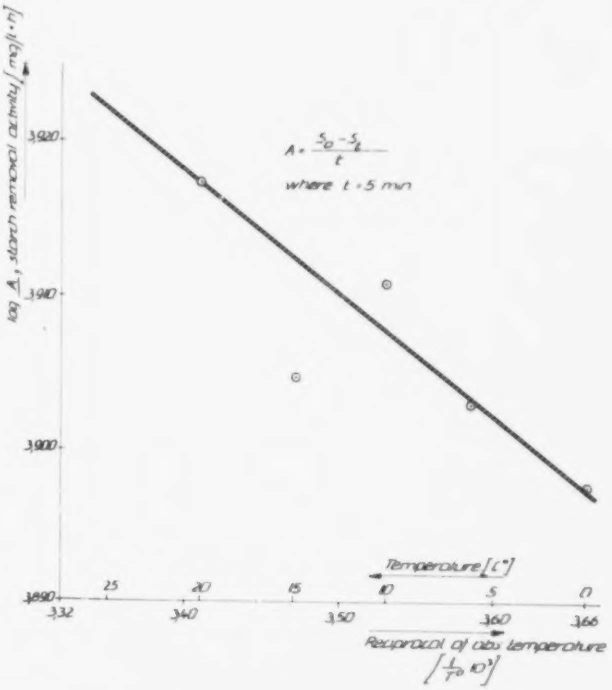


FIGURE 1 Temperature dependence of starch removal rate

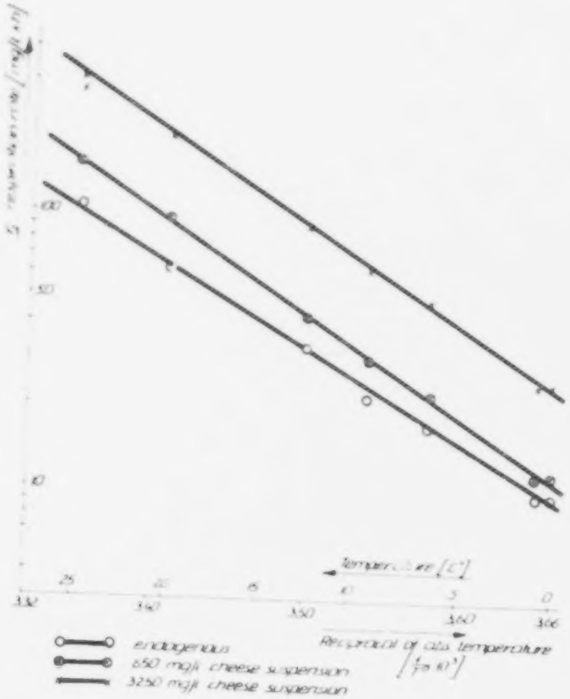


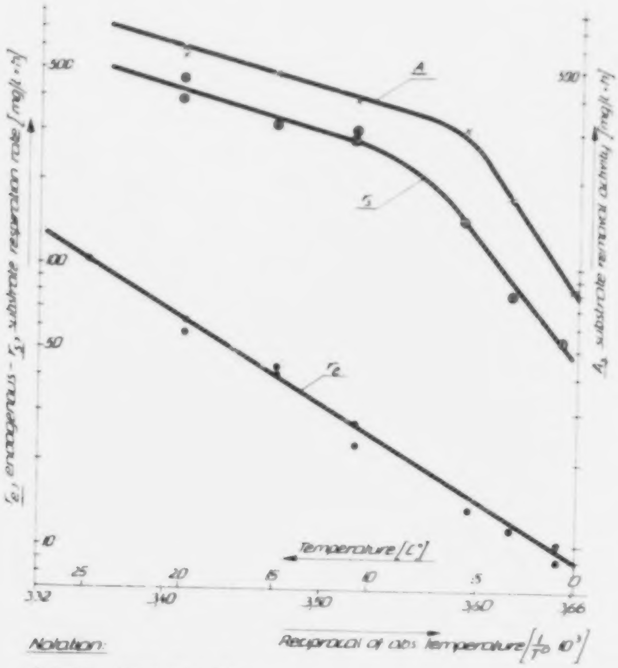
FIGURE 2 Temperature dependence of the biological oxidation rate of cheese suspension

Figure 6. Respiration at 20° C was arbitrarily taken as unity to make comparison easier. The activation energies were within the range of $\Delta H = 13{,}150 \pm 550$ cal.

All thermal dependence results obtained in this study together with some representative data from the literature are included in Table 2.

In Figure 7, the stoichiometric ratios of substrate oxidation, (g O_2/g substrate) obtained with STOD measurements are plotted versus temperature.

For acetate, this ratio seems to remain unaffected by temperature. The value of 0.4 g O_2/g acetate is in good agreement with the results of Warburg's measurements and data of other authors. For phenol, the stoichiometric ratio ranged from 0.7 to 0.9, the smaller figure being valid at 0° C.



Notation:

1, Acetate load 10 mg/l

2, Activated sludge grown on municipal waste

FIGURE 3 Temperature dependence of the biological oxidation rate of acetate

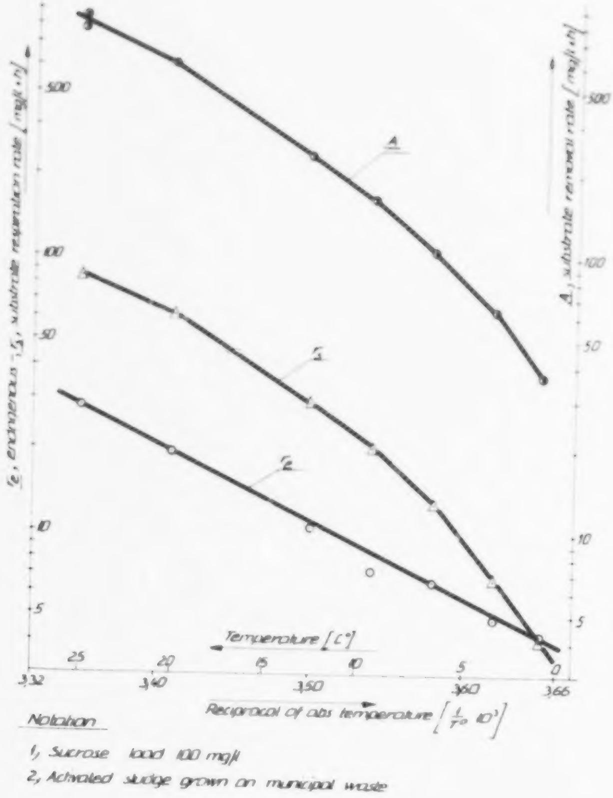


FIGURE 4 Temperature dependence of the biological oxidation rate of sucrose

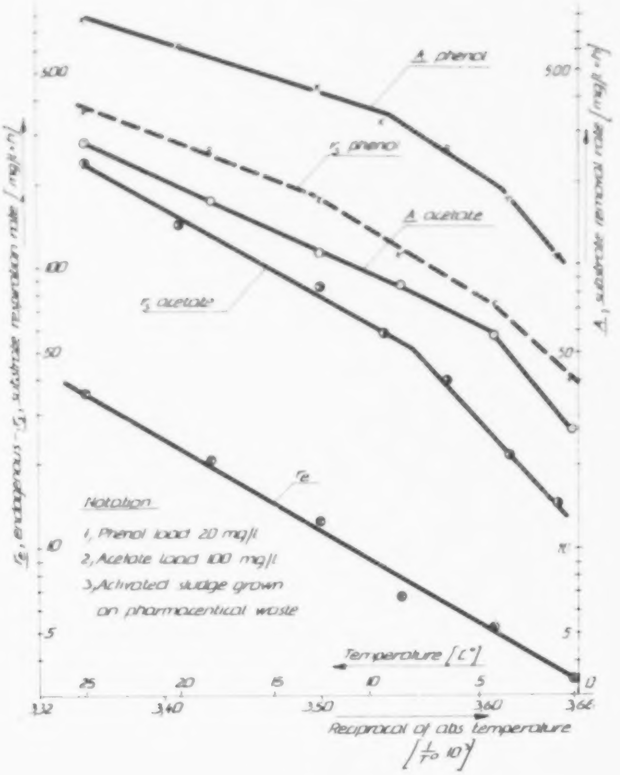


FIGURE 5 Temperature dependence of the biological oxidation rate of phenol and acetate, respectively

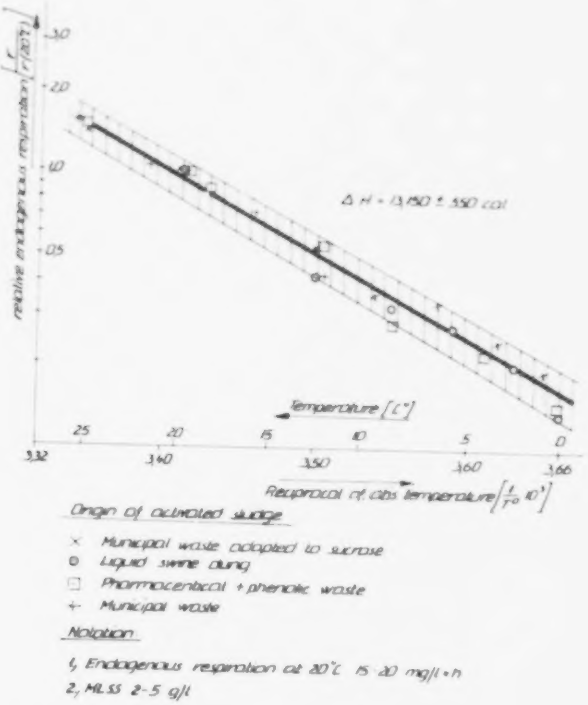


FIGURE 6 Temperature dependence of the endogenous respiration of four different activated sludge samples

DISCUSSION

The thermal dependence data were in good agreement with those of other authors. Johnson et al. (1957) statistically analyzed the available body of experimental data comprising a broad variety of biological processes. They found that two distinct peaks exist in the frequency of occurrence of ΔH values: one between 11,000 - 13,000 cal, (from the results reported herein, the ΔH values of dissolved substrates above 10° C - and perhaps, endogenous respiration - belong in this group), the other between 15,000 - 18,000 cal, where the removal of dissolved substrate belongs, at temperatures lower than 5° C.

According to the Crozier theory, substrate metabolism cannot be described with one simple Arrhenius equation, if it is assumed to be a complex process comprising a series of coupled biochemical reactions. At different temperatures there would be different "master reactions" characterized by different activation energies. (The reaction determining the overall reaction rate is called "master reaction" by Johnson et al., 1957). The temperature value, where there is a break

TABLE 2

Temperature Dependence of The Rate of Metabolism For Activated Sludge

Metabolism	Temperature range, C°	ΔH cal	Q_{10}	θ	Source
1. Adsorptive removal rate of starch	0 - 25	364	1.022	1.000	Fig. 1
2. Biological oxidation of caseine suspension	0 - 25	17.000	2.66	1.103	Fig. 2
3. Removal rate of acetate	0 - 10	7.200	1.51	1.042	Fig. 3
	10 - 25	33.700	6.90	1.214	Fig. 3
4. Substrate respiration with acetate	0 - 10	7.200	1.51	1.042	Fig. 3
	10 - 25	37.000	8.42	1.242	Fig. 3
5. Removal rate of sucrose	0 - 10	-	5.80	1.192	Fig. 4
	10 - 20	-	2.86	1.110	Fig. 4
6. Substrate respiration with sucrose	0 - 10	-	6.50	1.206	Fig. 4
	10 - 20	-	2.72	1.105	Fig. 4
7. Removal rate of phenol	0 - 10	-	3.87	1.131	Fig. 5
	10 - 20	-	1.72	1.056	Fig. 5
8. Substrate respiration with phenol	0 - 10	-	3.18	1.134	Fig. 5
	10 - 20	-	2.0	1.072	Fig. 5
9. Endogenous respiration (4 different sludges)	-	13.150	2.15	1.078	Fig. 6
10. Respiration of activated sludge	0 - 20		2.25	1.085	Eckenfelder, 1966
	0 - 25	-	2.04	1.074	Wuhrmann, 1964
11. Rate of nitrification	-		3.81	1.143	Downing, 1968
12. O_2 - consumption rate when determining BOD	4 - 20		3.55	1.135	Eckenfelder, 1966
	20 - 30	-	1.72	1.056	Eckenfelder, 1966

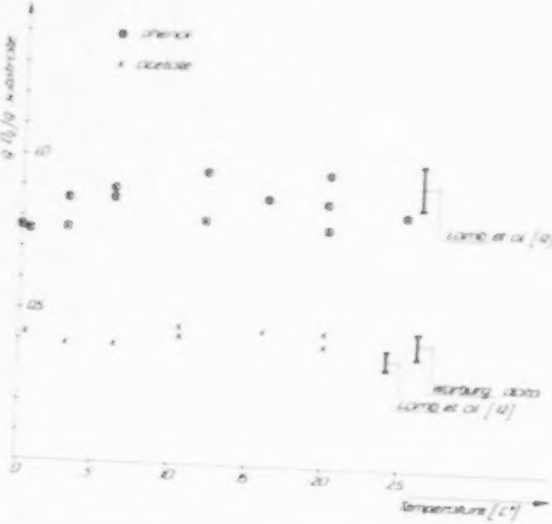


FIGURE 7 **Temperature dependence of substrate oxidation oxygen demand**

on the Arrhenius plot, is called the "critical temperature", where two master reactions are supposed to "switch over." In Figure 3, a sharp break at 7° C can be observed on the Arrhenius plots of both acetate removal and substrate respiration, and with all dissolved substrates, a more or less pronounced "bend" can be seen in the range of 5° C 10° C.

It is known that dehydrogenases have ΔH values of 11,200 calories, and cytochromes (respiration enzymes underlying cyanide poisoning) have ΔH = 16,000, but the data is too fragmentary to conclude that above 10° C the master reaction is dehydrogenation, and at 0° C, cytochrome-catalized respiration.

The Arrhenius plot of the metabolic reactions of activated sludge is similar to those of psychrophylic bacteria that exert a technologically utilizable metabolic activity even at 0° C (Ingraham, 1962).

Strikingly, the biooxidation of adsorbed food readily follows the Arrhenius law. According to present knowledge, adsorbed food must be hydrolyzed and transformed into dissolved compounds prior to biological utilization. Considering this, it remains an open question why caseine oxidation shows a temperature dependence different from the dissolved substrates.

Endogenous respiration also does not show any "critical temperature value." It seems plausible that endogenous metabolism has only one master reaction in the 0° C - 25° C temperature range. Endogenous "basic" metabolism has to be very stable because it is more important from the point of view of survival and resistance against adverse environmental conditions than substrate metabolism. It may be assumed that endogenous metabolism, as a well-stabilized biochemical reaction chain, has only one "master reaction", consequently behaving as one simple reaction.

The amount of substrate oxygen demand changes little with temperature. This is in fair agreement with the findings of others (Verrimmen et al., 1967 and Lamb et al., 1964) (Fig. 7). This means that in the instances studied only the reaction rate, but not its stoichrometric constants, has changed with the temperature. Similar conclusions were drawn from BOD curves recorded at different temperatures (Gotaas, 1948; Wilderer et al., 1969; Chia, 1969).

Final conclusions concerning thermal dependence of the activated sludge process can be drawn only from the continuous process. The calculation of K', according to Equation 12, is possible only if a large collection of data is at hand. The S_o/S_e ratio usually has a great fluctuation, and the data must be treated statistically (Wuhrmann, 1964).

CONCLUSIONS

1. Endogenous metabolism follows the Arrhenius law, while the metabolism of dissolved substrate does not. An explanation to this is sought in the Crozier theory, supposing that at different temperatures, different master reactions (having different activation energies) are the "bottlenecks" of the biological reactions.

2. The adsorptive mechanism of activated sludge also has a temperature dependence, but this amounts to less than 10% of the thermal dependence of biological processes.

3. The value of grams-oxygen consumed per gram substrate oxidized was found practically constant in the range of 0° - 25° C.

4. Temperature dependence of continuously working activated sludge plants is usually less than would be expected from non-steady state measurements. From the possible explanations of this phenomenon, the Eckenfelder theory of partial anaerobiosis within the flocs was mentioned. Further, it should be kept in mind that most wastes contain colloids, the removal rate of which does not depend on temperature. A great difference may be encountered in the thermal dependence of treatment efficiency, depending on whether the waste contains colloids or dissolved compounds, in favor of the former.

5. A great importance has to be attributed to heat storage in water bodies underlying biological treatment; a system with long detention time may not be economic, because the thermal reserve - consequently, higher removal rate - of a short detention time system may compensate for the long detention time with heat loss in a total oxidation plant.

APPENDIX

Interdependence Between The Thermal Coefficients of ΔH, θ, and Q_{10}

From the Arrhenius equation, Equation 4, it can be written:

$$\log \frac{A_2}{A_1} = \frac{\Delta H}{2.3\,R} \cdot \frac{T_2 - T_1}{T_2 \cdot T_1} \qquad \text{(A-1)}$$

From the above equation, $R = 1.99$ Kcal/mol.K°, and $T_2 \cdot T_1$ has the approximative value of 85,850 in the temperature range of 0° C - 30° C. Thus it can be written:

$$\log \frac{A_2}{A_1} = \frac{\Delta H}{(2.3)(1.99)} \cdot \frac{T_2 - T_1}{85{,}850} = \frac{\Delta H}{394{,}910} \cdot (T_2 - T_1) \qquad \text{(A-2)}$$

Introducing the θ constant:

$$\log \theta = \frac{\Delta H}{394{,}910} = \Delta H\,(2.54)(10^{-6}) \qquad \text{(A-3)}$$

From A-2 and A-3 we obtain:

$$\log \frac{A_2}{A_1} = \log \theta\,(T_2 - T_1)$$

or

$$\frac{A_2}{A_1} = \theta^{(T_2 - T_1)} \qquad \text{(A-4)}$$

which is the same as Equation 9.

For the parameter Q_{10}, observing definition according to Equation 10, it can be written:

$$\log \frac{A_{T+10}}{A_T} = \log Q_{10} = 10 \log \theta \qquad \text{(A-5)}$$

From Equations A-3 and A-5 it can be set up for the three parameters:

$$\log Q_{10} = 10 \log \theta = \Delta H\,(2.5)(10^{-5}) \qquad \text{(A-6)}$$

which is in accordance with Equation 11.

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TEMPERATURE EFFECTS ON BIOLOGICAL WASTE TREATMENT PROCESSES

W. Wesley Eckenfelder, Jr. and A. J. Englande

INTRODUCTION

Variations in liquid temperature affect biological waste treatment processes in two interdependent ways. First, the oxygen utilization rate which reflects the energy transfer of the process has a defined temperature relationship. This reaction temperature dependency is reflected by a temperature coefficient θ. Secondly, in any biological system a relationship will exist between the oxygen transfer rate to the biomass and the oxygen utilization rate of the biomass. It is therefore necessary that θ also consider the effect of temperature on the oxygen transfer rate. The absolute temperature effect on the performance of a particular process will therefore be the resultant of these two effects.

In order to estimate the thermal influence on a process, a modification of the Van't Hoff-Arrhenius equation is usually used:

$$K_T = K_{20^\circ C}\,\theta^{(T-20)} \tag{1}$$

It should be noted that this relationship is valid only within specific limits. A lower limit is imposed by retardation of bacterial activity for mesophilic organisms as the temperature approaches freezing. Relatively high reaction rates may still exist at very low temperatures for psychrophilic organisms The rate of the biological reaction rates will increase in accordance with Equation 1 with temperature to an optimum value for most aerobic systems. Further increases in temperature result in a decreased rate for mesophilic organisms. Maximum biodegradation by thermophilic organisms, however, will be obtained over a temperature range of 35° C to 65° C.

The effect of temperature on biological processes is the resultant of several factors which consider the type and distribution of microorganisms, the type of process and the design and operation of the system.

Bacteria can be broadly classified into three categories as to their response to temperature, namely psychrophilic, mesophilic and thermophilic. Each of these categories have an effective temperature range. By far, the majority of wastewater treatment processes function in the mesophilic range, although it has been demonstrated that some processes in northern climates function in the psychrophilic range. A few systems have been designed and operated in the thermophilic range. Probably, the most responsive factor relating temperature effect to process performance is the food/microorganism ratio (F/M). At high F/M values the biomass may be filamentous or dispersed and θ will be high indicating a direct temperature effect on each organism. By contrast, at lower F/M ratios the biomass is flocculated and diffusional mechanisms into the floc become significant. More of the floc is aerobic at lower temperatures so that a greater aerobic biomass at lower temperatures is capable of stabilizing almost the same quantity of organic matter as a smaller more active biomass at higher temperatures. It is because of this that the activated sludge process at conventional loadings (F/M 0.6) and trickling filters yield a much lower temperature coefficient, θ, than those processes using primarily dispersed growths such as the aerated lagoon. It is probable that the mixing intensity in the process which affects floc dispersion also influences the coefficient θ. Temperature effect on the various processes are shown in Figure 1.

In order to adequately predict the magnitude of the temperature influence on a system's efficiency, an appropriate θ value must be employed. The temperature coefficient will vary depending primarily on the nature of the process under consideration. This paper discusses θ for various biological process relationships. The processes considered include waste stabilization ponds, aerated lagoons, and activated sludge.

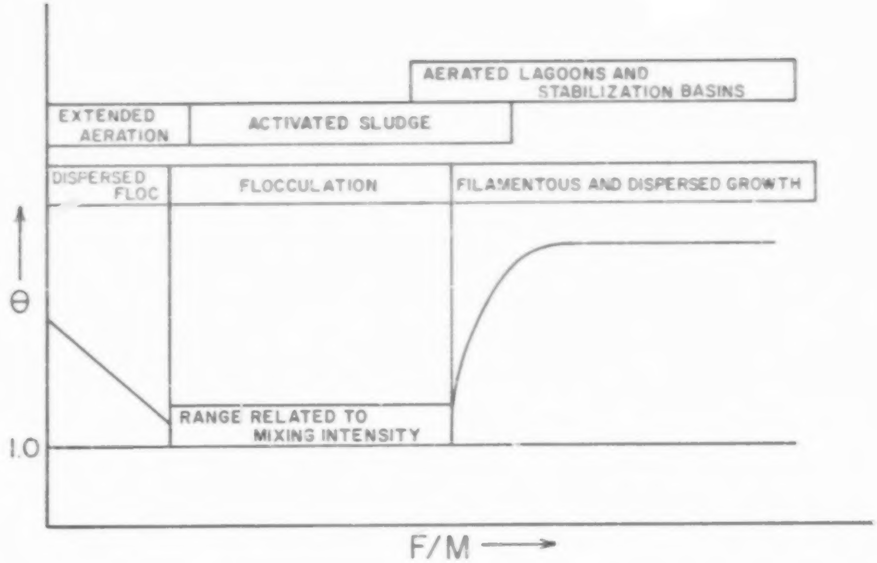
AERATED LAGOONS AND STABILIZATION BASINS
EXTENDED AERATION
ACTIVATED SLUDGE
DISPERSED FLOC
FLOCCULATION
FILAMENTOUS AND DISPERSED GROWTH
θ
RANGE RELATED TO MIXING INTENSITY
1.0
F/M

FIGURE 1 **Relationship between temperature coefficient, θ, and food to microorganism ratio, F/M**

DETERMINATION OF θ

The temperature influence on a waste treatment process is usually reported as a change in the percent removal of BOD. The temperature coefficient θ which reflects the change in reaction rate can be computed by graphical methods. Alternative calculations may be employed depending on the type of data available and the type of process used.

Figure 2 illustrates one method of analysis which is developed as follows for completely mixed aeration systems:

The BOD removal relationship can be defined as

$$\frac{S_o - S_e}{S_e} = kX_v t \tag{2}$$

in which

S_o and S_e are the influent and effluent BOD concentrations respectively

X_v is the aeration volatile solids concentration

t is the aeration time

k is the mean reaction rate coefficient

and

$$\frac{S_o - S_e}{S_o} = \% \text{ removal } (R) \tag{3}$$

Combining Equations 2 and 3 one obtains:

$$\frac{R}{1 - R} = kX_v t \tag{4}$$

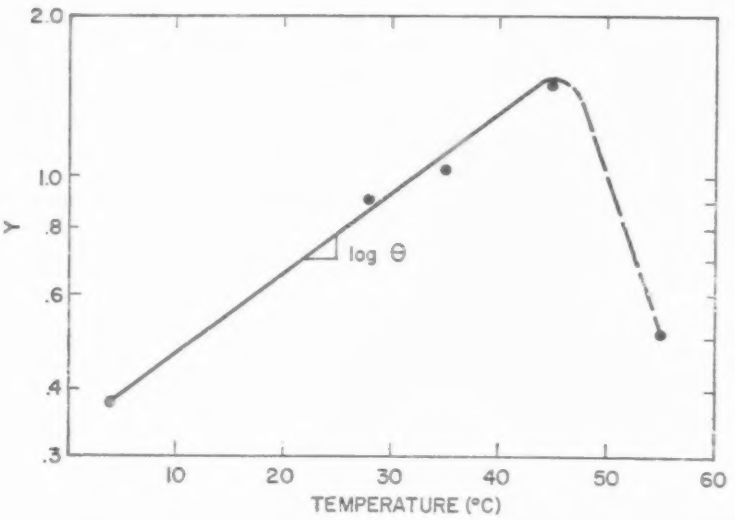

FIGURE 2 Temperature function for activated sludge (Hunter et al., 1967)

Let the left side of Equation 4 equal Y. Then, if X_v and t remain constant, k_T is proportional to Y.

One can rewrite the equation $k_T = k_{20}\theta^{T-20}$ in the form

$$Y = G \cdot \theta^{T-20} \tag{5}$$

where G is a constant;

thus

$$\log Y = \log G + (T-20) \log \theta \tag{6}$$

As shown in Figure 2, θ is computed from the slope of a semilog plot of Y versus temperature for a constant t and X_v.

The temperature coefficient for aerated lagoons where the detention time varies can be computed from Equation 1:

$$\log k_T = \log k_{20} + (T-20) \log \theta \tag{1a}$$

A semilog plot of percent BOD remaining versus detention time will yield a slope representative of the reaction rate k. In accordance with Equation 1a, a plot of k versus temperature (as shown in Figure 4) will yield the coefficient θ.

Values for the reaction rates may also be obtained by direct calculation using Equation 2, and the coefficient θ computed.

WASTE STABILIZATION PONDS

Temperature has a significant influence on waste stabilization pond efficiency. Both photosynthetic oxygen production and biological degradation rates are greatly affected by temperature variations. While optimum photosynthetic activity is maintained at approximately 20° C, the upper and lower limits appear to be about 35° C and 3° C respectively.

A value for θ of 1.072 was obtained by Herman and Gloyna (1958) for a temperature range between 3° C and 35° C. Suwannakarn and Gloyna (1964) found θ equal to 1.085 for a synthetic soluble sewage between 9° C and 35° C. Oswald (1966) found a temperature coefficient of 1.075 for domestic sewage between temperature limits of 10° C and 18° C. All the reported values are in close agreement, providing an average θ value of 1.077.

ACTIVATED SLUDGE

Table 1 summarizes θ values computed from data available in the literature for activated sludge processes. Temperature coefficients range from 1.0 to 1.041. The value of 1.076 computed from Wuhrmann (1966) is the probable result of a dispersed growth due to the very high food-microorganism ratio of 2.03. Data reported by Hunter et al. (1967) are shown in Figure 2. A value of θ was determined to be 1.035 for the synthetic sewage over a temperature range of 4° C

TABLE 1

Temperature Coefficient Evaluation

Activated Sludge

Temp. Range °C	F/M**	MLVSS (mg/l)	S_o (mg/l)	Substrate	Batch or Continuous	Detention Time (hr)	θ	Source
4 - 45	0.58	1,600*	623	Synthetic Sewage (dry dog food meal)	Batch (bench scale)	16	1.035	Hunter et al. (1967)
5 - 30	0.44	1,100	435	Slurried dog food meal	Continuous (bench scale)	12	1.037	Ludzack et al. (1961)
	0.81	1,100	435	Dog food meal with dextrose & gelatin			1.041	
	0.45	1,870	750	Phenol			1.016	
26 - 37	0.57	3,200*	229	Kraft black liquor	Continuous (bench scale)	3	1.006	Carpenter et al. (1968)
9 - 17	2.03	480*	108	Domestic Sewage	Continuous (pilot plant)	2.67	1.076	Wuhrmann (1966)
	0.43	2,640*	124				1.0	
	0.22	4,800*	115				1.0	
10 - 25	0.74	800*	248	Domestic Sewage	Batch (bench scale)	10	1.015	Sawyer (1940)

to 45° C. Further increases in temperature resulted in a sharp decrease in process efficiency. This data could be interpreted to show a detrimental effect of temperature below 20° C and above 45° C. A lower optimal temperature was found by Carpenter et al. (1968) as 37° C. An activated sludge plant at a West Virginia Pulp and Paper Company plant showed a decrease in process efficiency at temperatures in excess of 40° C. The activated sludge process appeared to function satisfactorily at temperatures as low as 4° C.

TABLE 2

Temperature Coefficient Evaluation

Aerated Lagoons

Temp. Range °C	F/M	MLVSS (mg/l)	S_o (mg/l)	Substrate	Batch or Continuous	Detention Time (days)	θ	Reference
10 - 30	--	--	--	Cotton Textile Waste	Continuous (pilot plant)	2.6 - 5	1.035	Sawyer (1940)
13 - 20	2.51	106*	266	Domestic Sewage	Continuous	8.6	1.046	Eckenfelder (1971)
2 - 30	2.50	80*	200	Kraft	Continuous for 2.5 & 5 day detention	2.5 - 10	1.031	Carpenter et al. (1968)
				Neutral sulfite semichemical			1.046	
				Acid sulfite	Batch	10	1.039	
				Roofing felt			1.040	
				Board mill	(bench scale)		1.031	
				Avg. of all 5			1.035	
				<10° C			1.058	
				>10° C			1.026	

* Assumed MLSS = 50% influent BOD

Assumed MLVSS = 80% MLSS

AERATED LAGOONS

Pertinent descriptives and computed temperature coefficients for the aerated lagoon process are summarized in Table 2. A range of θ from 1.026 through 1.058 was observed.

The variation in BOD removal characteristics over a temperature range of 10° C - 30° C from continuous treatment studies by Sawyer (1966) is shown in Figure 3. The temperature coefficient from these data is 1.035.

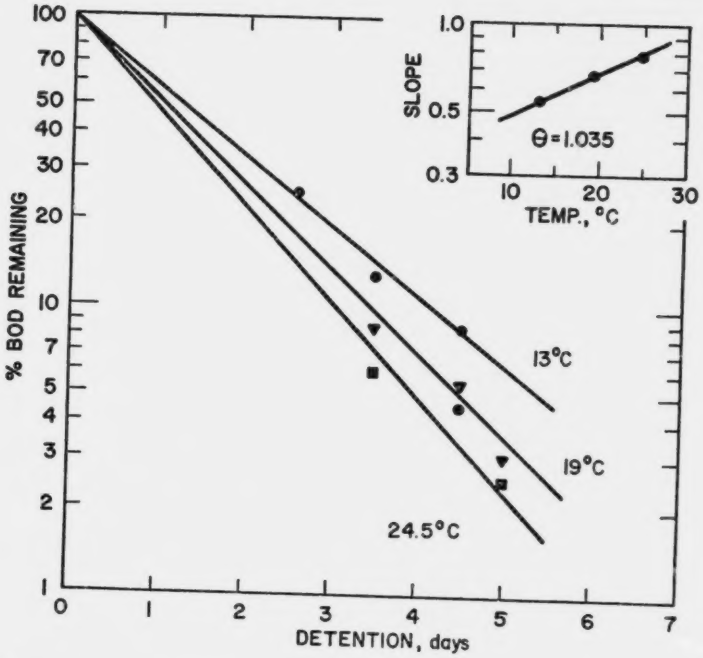

FIGURE 3 Temperature function for aerated lagoon (Sawyer, 1966)

Carpenter et al. (1968) conducted an extensive study of pulp and paper mill wastes and calculated an average θ of 1.035 for five different wastes over a temperature range of 2° C - 30° C. Figure 4 shows the graphical solution for θ. It is significant to note that two temperature coefficients might better describe the presented data than the one reported. The same trend was observed for each waste indicating a θ equal to 1.026 and 1.058 for temperature limits of 10° C - 30° C and 2° C - 10° C respectively.

Sawyer (1966) and Carpenter et al. (1968) conclude that aerated lagoons with short detention times will be extremely sensitive to temperature change. This effect is dampened by retention periods in excess of five days.

There have been recent reports of a relatively small temperature effect on aerated lagoons treating domestic sewage in northern climates. Two factors must be considered in interpreting such data. First, the major portion of the BOD in domestic sewage is present in suspended and colloidal form and a primary removal mechanism is adsorption and flocculation. These mechanisms are relatively insensitive to change in temperature. Secondly, the retention periods in most aerated sewage lagoons are long and changes in removal due to temperature are wasted. It is the writer's contention that further studies on temperature effects of these systems are needed before valid conclusions can be drawn.

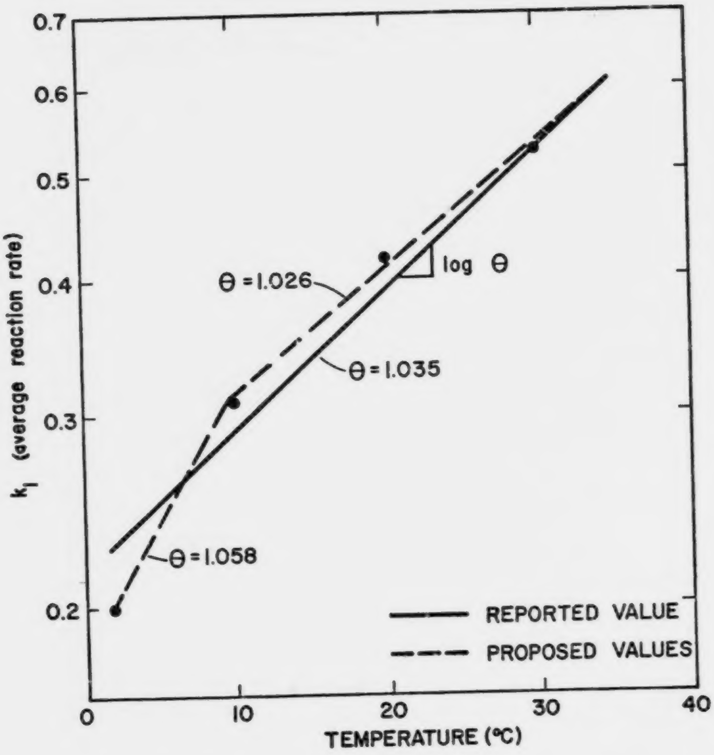

FIGURE 4 Derivation of temperature coefficient for aerated lagoon

DISCUSSION

The summary presented in Table 3 indicates the appropriate θ and corresponding temperature range for the various aerobic biological processes employed in wastewater treatment. Variations in the temperature coefficient can be rationalized by considering the inherent difference in process operation.

The parameter most influential in determining process temperature sensitivity is the food to microorganism ratio (F/M). The characteristics of the biomass will be dictated in major part by F/M. At very high values of F/M the biomass is dispersed. At intermediate organic loading levels dispersed filamentous growths may develop. At low loading levels, flocculation occurs, resulting in the formation of gelatinous flocs containing millions of individual organisms. Under starvation conditions floc dispersion will result. In the activated sludge process an F/M ratio approximately equal to 0.5 results in a flocculated biological growth. BOD removal and oxidation are obviously affected by such growth. The amount of oxidation depends on diffusion of oxygen into the biological floc, where it is subsequently utilized by the microorganisms. At conventional mixing intensities, relatively large flocs are generated. The portion of these flocs that are aerobic depends upon a balance between oxygen diffusion into the floc from the surrounding liquid and the oxygen consumption by the organisms contained within the floc. At low temperature, a low oxygen utilization rate permits diffusion of oxygen to a greater depth in the floc and therefore a large portion of the floc is aerobic. At high temperatures, the increased respiration rate depletes the oxygen rapidly, and only a small portion of the floc is aerobic. It can be postulated that a large mass of organisms at a low respiration rate (winter) achieves the same degree of oxidation as a small mass at a high respiration rate (summer), and hence the computed coefficient θ may be close to 1.0. At high mixing intensities in the aeration basin, the smaller floc sizes may be fully aerobic under all temperature conditions and the coefficient θ will increase. At the higher F/M ratios, filamentous or dispersed growths will exhibit a high temperature dependency as shown by Wuhrmann (1966).

TABLE 3

Summary Table

Process	θ Range	Temperature Range ° C	Reference
Stabilization Pond	1.072 - 1.085	3 - 35	*
Activated Sludge	1.0 - 1.041	4 - 45	*
Aerated Lagoon	1.026 - 1.058	2 - 30	*
Trickling Filter	1.035	10 - 35	Howland (1958)
Aerobic - Facultative Lagoon	1.06 - 1.18	4 - 30	Eckenfelder (1970)
Anaerobic Lagoon	1.08 - 1.10	5 - 30	Dietz et al. (1966)
Extended Aeration	1.037	10 - 30	National Sanitation Foundation (1966)

* This paper

It would therefore appear that the effect of temperature on the activated sludge process is related both to the loading level (F/M) and to the intensity of mixing in the process. Trickling filters are analogous to activated sludge except that oxygen diffusion into the film is uniplaner which results in a θ value in the order of 1.035.

Waste stabilization ponds, aerated and aerobic-facultative lagoon and anaerobic pond processes operate at lower solids levels and/or higher F/M. Recent data have shown that the biomass does not effectively flocculate at concentration levels below approximately 500 mg/l. Therefore, within these systems, the biomass is dispersed and hence the process is more directly affected by variations in temperature.

High solids concentrations and long retention periods contribute to a low F/M ratio (approximately 0.12) and a dispersed floc in the extended aeration process. The value of θ would therefore be expected to be higher than that for conventional activated sludge.

The results for various processes are summarized in Table 3.

CONCLUSIONS

1. The effect of temperature upon process efficiency can be formulated in terms of a temperature coefficient, θ. θ can be obtained graphically by variations of the modified Van't Hoff-Arrhenius equation.

2. Temperature effects are minimal on activated sludge process performance as compared to other treatment systems. The low sensitivity of the activated sludge process is primarily due to the F/M ratio which results in a flocculated biological growth.

3. There is some evidence that θ will increase at temperatures less than 10° C.

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EVALUATION OF AERATED LAGOONS AS A SEWAGE TREATMENT FACILITY IN THE CANADIAN PRAIRIE PROVINCES

Archie R. Pick, George E. Burns,
Dick W. Van Es and Richard M. Girling

INTRODUCTION

Metropolitan Winnipeg has a population of 500,000 and is located at latitude 49° 45′ N, longitude 97° 15′ W. The climate is of the continental type, with an annual temperature of 36.50° F, the coldest month is January with an average temperature of -20° F, the warmest month is July with an average temperature of 67° F above, and the average frost-free period (32° F) is 115 days. The average winter snowfall is 51 inches.

In 1967, the Metropolitan Corporation of Greater Winnipeg undertook a two-year study of aerated lagoons, because there was little documented information on aerated lagoons operating under Canadian prairie conditions. In order to assess the applicability of this process for the treatment of domestic wastes, three pilot aerobic-anaerobic aerated lagoons were constructed by the Corporation.

During the summer and fall of 1967 the pilot lagoons were constructed in the corner of an existing stabilization pond. The sewage treated was domestic sewage from a separate system. The three aeration systems installed were:

Air-Aqua*
Mechanical Surface Aerator**
Air-Gun***

The work was supported by Public Health Research Grant 606-7-167 of the Department of National Health and Welfare, Canada.

DESCRIPTION OF THE PILOT LAGOONS

Each system was designed to treat a flow of 0.5 Imgd. The general arrangement of the systems is shown on Figure 1 and the design data is summarized in Table 1.

Air-Aqua

This system operates on the diffused air principle. A 30-HP compressor supplies air to polyethylene tubes laid in a tapered grid on the cell bottom. The system has a 30-day retention time, an operating depth of 10 feet and is divided into two cells, operating in series. 10% of the effluent is returned to the inlet for seed.

* As manufactured by Hinde Manufacturing Limited, Hamilton, Ontario.

** Equipped with Lightnin Aerators as manufactured by Greey Mixing Equipment Limited, Toronto, Ontario.

*** As manufactured by Aero-Hydraulics Corporation, Montreal, P.Q.

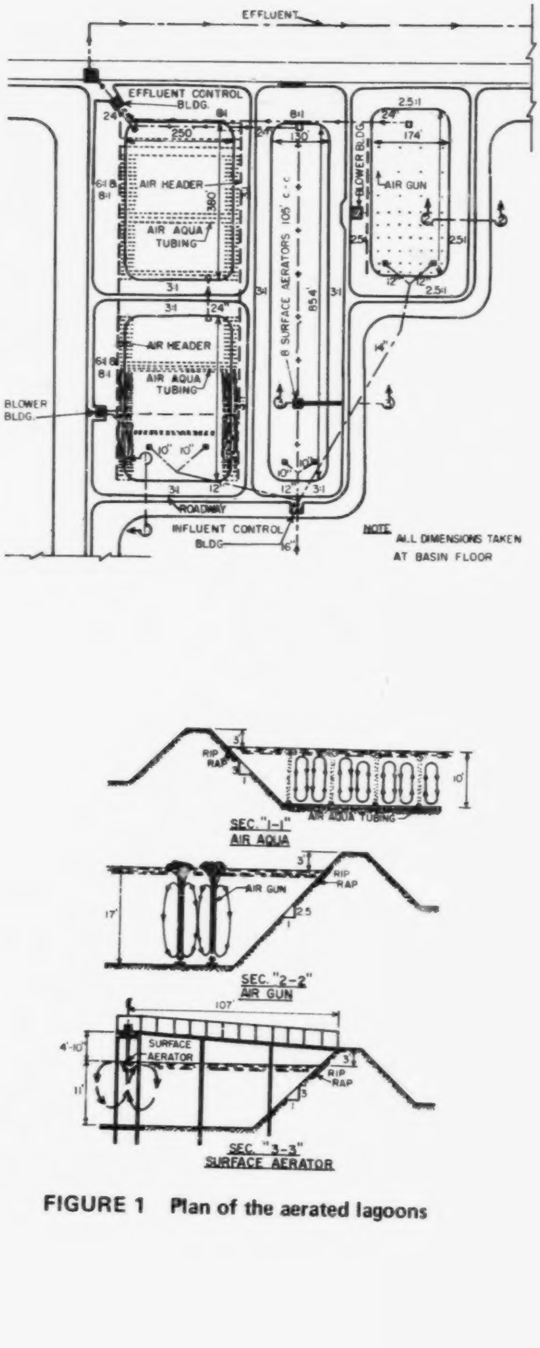
EFFLUENT
EFFLUENT CONTROL BLDG.
24"
8:1
250'
8:1
24"
130'
2.5:1
24"
174'
BLOWER BLDG.
AIR GUN
6:1 & 8:1
AIR HEADER
380'
AIR AQUA TUBING
8 SURFACE AERATORS 105' c-c
854'
2.5:1
2.5:1
12"
12"
2.5:1
3:1
3:1
3:1
3:1
24"
AIR HEADER
6:1 & 8:1
AIR AQUA TUBING
14"
BLOWER BLDG.
10"
10"
10"
10"
12"
12"
3:1
3:1
ROADWAY
INFLUENT CONTROL BLDG
16"
NOTE
ALL DIMENSIONS TAKEN AT BASIN FLOOR
3'
RIP RAP
3
1
10'
AIR AQUA TUBING
SEC. "1-1"
AIR AQUA
3'
RIP RAP
AIR GUN
17'
2.5
1
SEC. "2-2"
AIR GUN
107'
4'-10"
SURFACE AERATOR
11'
3'
RIP RAP
3
1
SEC. "3-3"
SURFACE AERATOR

FIGURE 1 Plan of the aerated lagoons

TABLE 1

Summary of Design Data

Item	Value and Units
Average Design Flow	0.5 Imgd
Influent 5-day BOD 20° C	250 mg/l
Influent Suspended Solids	180 mg/l
BOD Removal rate Coefficient (Base 10) k_1 at 0° C	0.13 per day
BOD Removal rate Coefficient (Base 10) k_1 at 20° C	0.50 per day
Temperature Coefficient θ (applied to k_1)	1.072 per ° C
Oxygen Utilization factor a' (lbs. oxygen required per lb. 5-day BOD removed)	1.50
Oxygen transfer ratio (ratio of O_2 transfer to waste to that of water) (α factor)	0.85
Saturation value of waste compared to H_2O (β factor)	0.95
Solubility of oxygen 20° C (780')	9.02 mg/l
Operating dissolved oxygen	2.00 mg/l
Effluent Temperature - winter	32° F
Effluent Temperature - summer	75° F
Influent Temperature - winter	48° F
Influent Temperature - summer	65° F
Mean ambient air temperature - winter	-16° F
Mean ambient air temperature - summer	+75° F
Treatment efficiency required	90%
Retention time - Air-Aqua	30 days
Retention time - Surface Aerator	20 days
Retention time - Air-Gun	20 days
Operating depth - Air-Aqua	10 ft.
Operating depth - Surface Aerator	11 ft.
Operating depth - Air-Gun	17 ft.
Volume - Air-Aqua	15×10^6 gal.
Volume - Surface Aerator	10×10^6 gal.
Volume - Air-Gun	10×10^6 gal.
Mixing requirements for surface aerators	0.016 HP/1000 gals.
Process loading - a) Design	
Process loading - - Air-Aqua	0.52 lbs. $BOD_5/1000\ ft^3/day$
Process loading - - Surface Aerator	0.78 lbs. $BOD_5/1000\ ft^3/day$
Process loading - - Air-Gun	0.78 lbs. $BOD_5/1000\ ft^3/day$
Process loading - b) Actual	
Process loading - - Air-Aqua	0.37 lbs. $BOD_5/1000\ ft^3/day$
Process loading - - Surface Aerator	0.55 lbs $BOD_5/1000\ ft^3/day$
Process loading - - Air-Gun	0.55 lbs. $BOD_5/1000\ ft^3/day$

Design Formulation:

$$t = \frac{E}{2.3\,k_1\,(100 - E)}$$

$$k_T = k_{20}\,\Theta^{(T-20)}$$

$$\text{lbs. } O_2 \text{ per day} = R_r = a' \text{ lbs. } BOD_5 \text{ removed/day}$$

Surface Aerator

This system consists of eight 20-HP aerators installed in series along the lagoon basin. The basin is sized for a 20-day retention time and has an operating depth of 11 feet. The raw sewage is fed in adjacent to the first aerator only.

Air-Gun

A combination of the diffused air and surface aerator principles is used for this system. There are 54 guns installed in a tapered grid pattern. A 40-HP compressor delivers air into an inverted siphon in the gun base, where a large bubble is formed which rises inside the gun, pushing water ahead and exploding at the surface. The system has a 20-day retention time and an operating depth of 17 feet.

RESULTS

Effluent Quality

The raw sewage concentration for the period January 1, 1968 to September 30, 1969 averaged 175 mg/l and 188 mg/l for BOD_5 and SS respectively. Corresponding effluent quality for the same 21 month period was:

	BOD mg/l	S.S. mg/l
Air-Aqua	37	34
Surface Aerator	38	39
Air-Gun	34	34

Figures 2 and 3 show these in detail.

The effect of cold weather on BOD_5 removal is evident when the winter of 1968-69 is compared to the summer of 1968 on Figure 2. The loss of efficiency during the summer of 1969 is attributed to sludge which will be discussed below.

The results were analyzed statistically for the 12 month period September 30, 1968 to September 30, 1969 (See Fig. 4 & 5). This period was selected as the most typical for continuous operation.

The median value of the effluent BOD for the three systems ranged between 39 and 38 mg/l. On 10% of the occasions the effluent was greater than or equal to 78 mg/l.

Suspended Solids removal remained reasonably constant over the 21-month period, with the exception of start-up, which can be attributed to initial erosion and suspension of materials from construction. On the 12-month basis the median SS in the effluent ranged between 21 and 23 mg/l and on 10% of the occasions greater than or equal to 48 mg/l.

Figures 6 and 7 illustrate results for temperature and D.O. respectively.

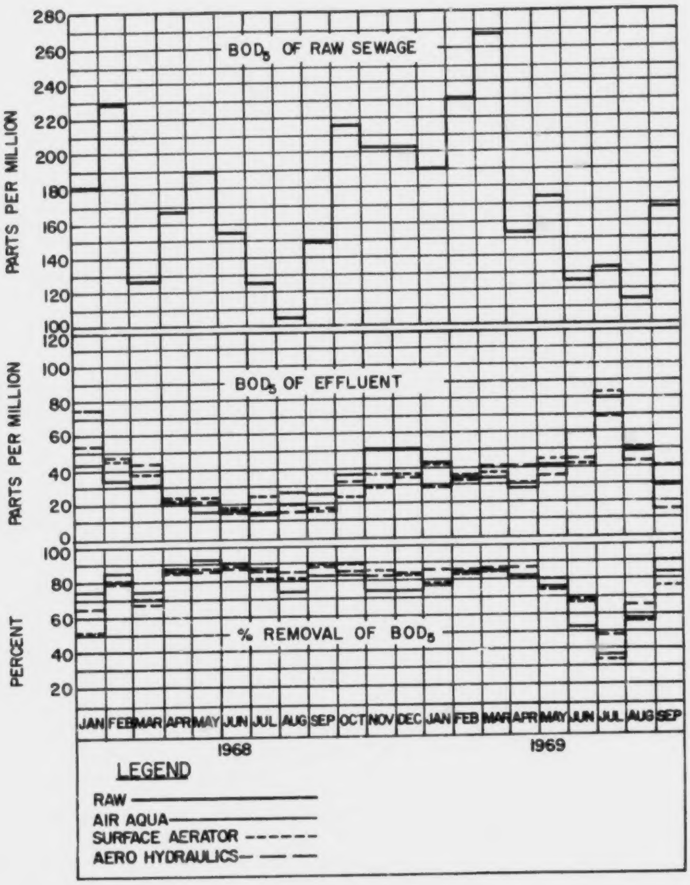

FIGURE 2 Lagoon performance (BOD$_5$)

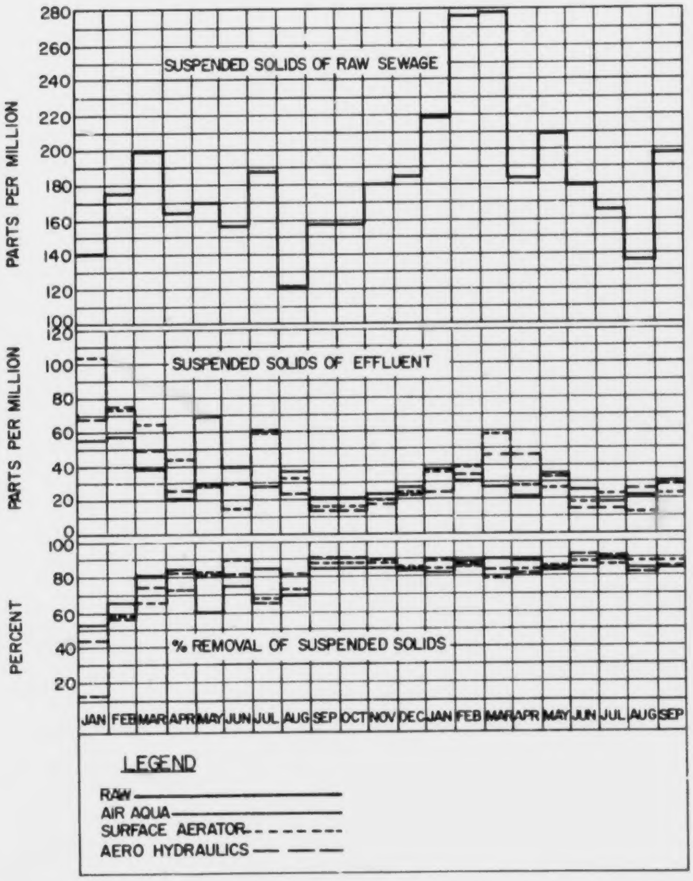

FIGURE 3 Lagoon performance (Suspended solids)

Nutrients

Figures 8 and 9 show the average performance of the aeration systems in the treatment of algal nutrients, phosphorus and nitrogen. The average percent removals over the test period were:

System	Nutrient Removal	
	Total Nitrogen (N)	Phosphate (PO_4)
Air-Aqua	12.6%	16.9%
Surface Aerator	14.6%	19.5%
Air-Gun	10.0%	23.3%

Although there was a reduction of total phosphorus through all systems, the orthophosphate concentrations in the effluent increased. The nutrient removal efficiencies were relatively low compared to results on ponds reported by others (Azzenso and Reid, 1966). The conventional activated sludge plant operated by the Corporation has removal rates of 36% and 45% for nitrogen and phosphate respectively. There was no appreciable difference in removal rates between summer and winter.

The aeration systems are basically equal in nutrient removal and are relatively ineffective.

Temperature

The effluent temperature follows the ambient temperature curve closely as it is almost independent of the raw sewage temperature (Fig. 6). There is a four-month period during the year when the effluent temperature is between 0° C and 1° C.

One of the concerns in design was the possible freezing of the Air-Gun cell, as little information was available on heat loss through ice cover. To keep the heat loss at a minimum, the surface area was reduced by making the side slopes steeper. Observations proved, however, that the cell had a built-in self-protection system. When the temperature rose to 10° F the ice melted on 25% of the cell, but as soon as the temperature dropped the cell covered with ice to conserve heat.

SLUDGE ACCUMULATION

The three aeration systems have shown a substantial build-up of bottom sludges. Samplings were conducted during July 1968, in the fall of 1969 and early in 1970. The accumulation of sludge in aerated lagoons has been recognized by others (Thimsen, 1965; Barnhart, 1965; Clark and Dostal, 1968); however, the significance of the accumulation of sludge deposits under climatic conditions similar to those experienced in the Canadian prairies has not been reported.

The rate of sludge accumulation in the Air-Aqua system has been estimated to be approximately 0.18 to 0.25 lbs. of dry solids per capita per day (based on population equivalents). The rate is comparable to the 0.21 lbs. of dry solids per capita per day pumped to digestion at an activated sludge plant operation by the Corporation.

Theoretical Sludge Accumulation

Theoretically, the overall digestion rate is sufficient to reduce the amount of volatile sludge accumulated to a relatively stable content each summer. After the first summer of operation an

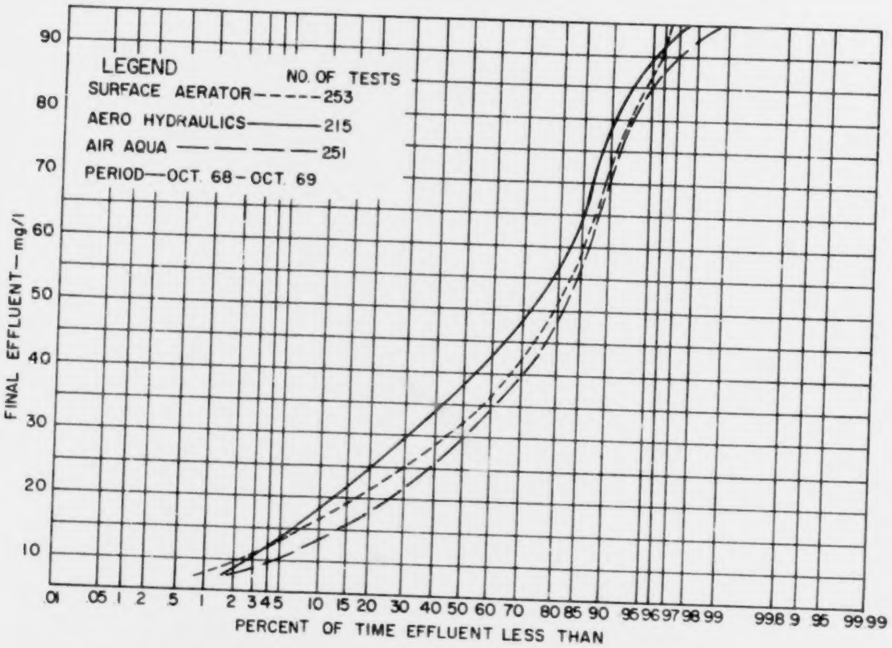

FIGURE 4 Variability of final effluent BOD_5 (Test lagoons on a one year basis)

annual cycle should occur with the sludge accumulation reaching a maximum in March-April and decreasing to a minimum level in August-September. The theoretical cycle is shown in Figure 10 (based on temperature-corrected anaerobic digestion rates).

Based on the accumulation of all suspended solids removed, and 0.45 pounds of solids synthesized per pound of BOD_5 removed, the daily sludge production would be 1,120 pounds. If the end products of anaerobic digestion had been removed from the systems, the amount of accumulation at the September 1969 sampling would have been an estimated 200,000 pounds (average daily accumulation of approximately 300 pounds).

Observed Sludge Accumulation

Based on actual surveys, the sludge accumulation (as dry solids) to September 5, 1969 was estimated as follows:

	Total Lbs.	Lbs/day	Lbs/θ/day
Air-Aqua Primary	740,000	1,120	0.21
Air Aqua Secondary	145,000	220	0.04
Air-Gun	580,000	930	0.18

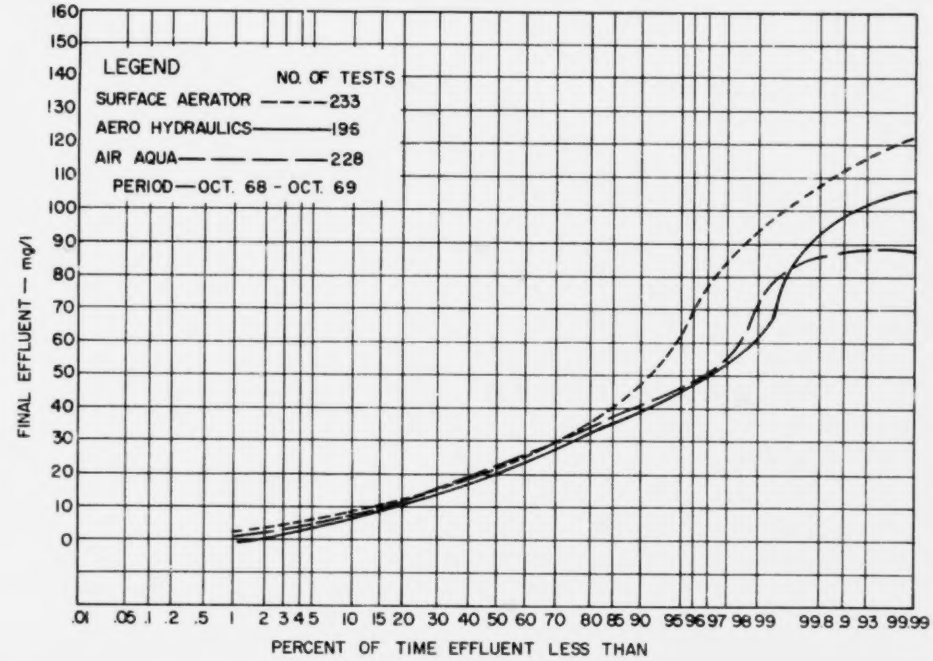

FIGURE 5 **Variability of final effluent suspended solids (Test lagoons on a one year basis)**

Sampling of the Air Gun had to be done through 15 feet of supernatant liquor and these figures are subject to confirmation. Only two samples were obtained in the Air-Aqua secondary, so the reliability of the 145,000 pound estimate is poor. Considering the methods of obtaining data, it was concluded that there was no apparent difference in the quantities of sludge between the systems, the significant point being that the accumulation was considerably greater than expected. An analysis of the sludge from the Air-Aqua system indicated a moisture content of 90% for the primary cell and 92% for the secondary cell, the volatile content was 55% for the primary and 45% for the secondary.

Effect of Sludge Accumulation on Aerated Lagoon Performance

During the months of May, June, July and August 1969, in all three systems, there were significant upward trends in the effluent BOD, as shown in Figure 11. All three systems showed similar relationships. This decline in BOD removal efficiency was accompanied by a trend to reduced dissolved oxygen concentrations in all cells. The reduced dissolved oxygen could be an "effect" caused by high oxygen demands imposed by the end-products of anaerobic decomposition of the sludge, or it could be a "cause" of higher effluent BOD due to an oxygen deficiency.

Some of the related factors that may account for the sludge accumulation and the effect on performance are:

a) Release of sludge digestion end products to the mixed liquor
b) Recycle of sludge by bacterial and algal synthesis in the mixed liquor
c) Relatively short period of higher rate anaerobic digestion
d) Insufficient air supply

The duration of the study has been insufficient to allow definition of the ultimate extent of sludge accumulation and loss of efficiency during the summer. However, based on the observations made, it appears that sludge is accumulating at the rate of approximately one ton of dry solids per Imgd treated.

To determine if an abnormal quantity of inert material was contained in the raw sewage or effluent a series of tests was conducted. The volatile content of the raw sewage was normal and showed no evidence of extraneous inert matter. Similarly, the effluent volatile solids were typical for biologically treated sewage.

Insofar as the higher BOD and reduced D.O. in the effluent is concerned, it is probably that

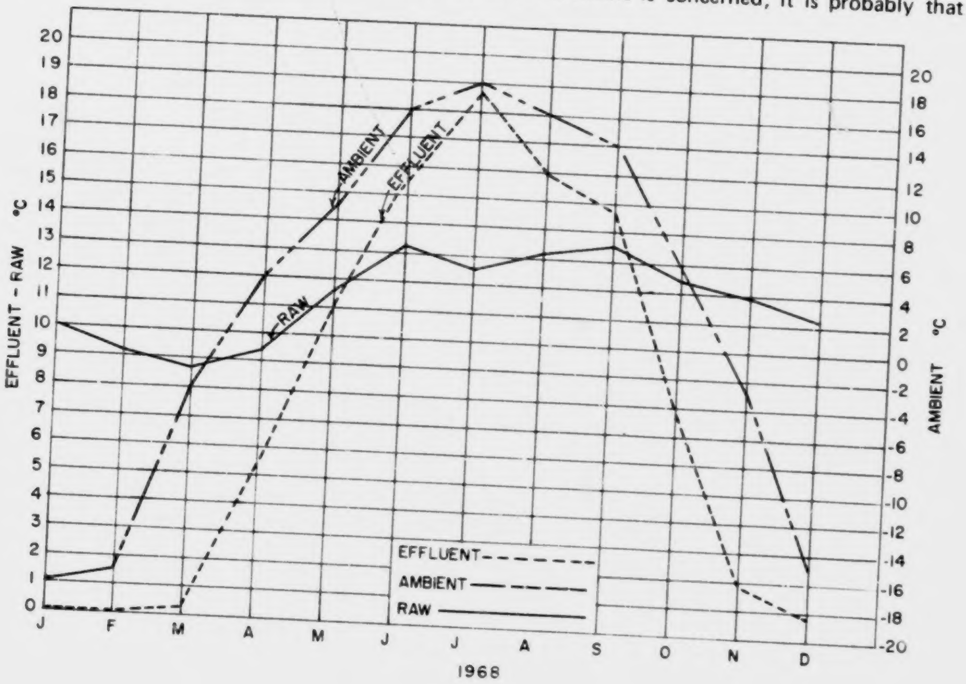

FIGURE 6 Temperature monthly average

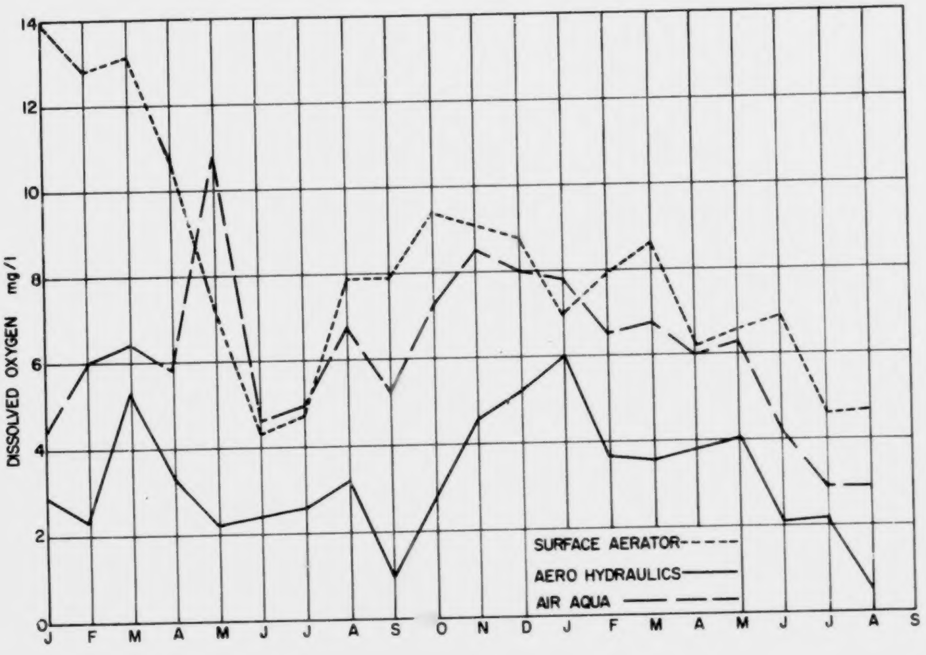

FIGURE 7 **Dissolved oxygen average monthly (372 readings per system)**

sufficient additional air can be added economically to handle the benthal demand, although some doubts exist because the Surface Aerator system sustained a similar loss of efficiency in spite of relatively higher D.O. concentrations. The major problem would appear to be the physical accumulation of the sludge, which will ultimately require removal and additional treatment.

Sawyer has described the problems encountered as being similar to experience with Imhoff tanks, with respect to removal and storage of BOD and solids during the winter months and the release of soluble BOD and nutrients as acid fermentation of accumulation sludge deposits develops. (Sawyer, 1970).

OPERATIONAL OBSERVATIONS AND PROBLEMS

Observations were made on mosquitoes, weeds, grease and scum odors, ice, foaming, grit, erosion and equipment operation. Plugging of the Air-Aqua tubing and the Air-Guns occurred. The problems were corrected and at the time of writing, both systems appear to be operating satisfactorily, although more time is needed to assess their ultimate reliability. Ice was a major problem with operation of the surface aerators; ice built up on the impeller shaft and support structure with subsequent freeze-up and stoppage. Although steps were taken to try and eliminate the ice problem, it was found that during the winter months only half of the surface aerators could be kept operating.

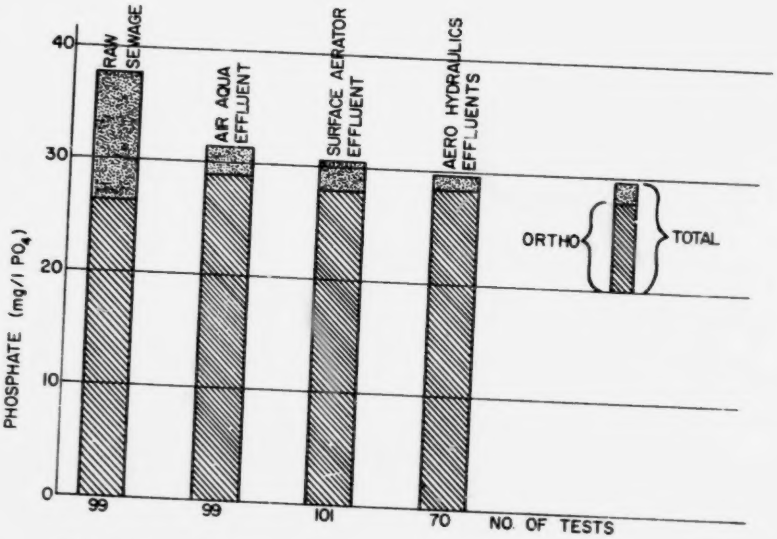

FIGURE 8 Test aerated lagoons average test results phosphates

Weed Control

No particular problems were encountered with the control of weed growth on the test cell banks. Scheduled grass cutting on the sodded banks and judicious application of weed control chemicals on the rip-rapped sections of the banks were carried out to control this nuisance with good success.

Mosquito Control

Although at times the mosquito population appeared to be large, actual counts taken showed that the lagoon area had one of the lowest mosquito counts of the entire Metropolitan Area. Here it should be noted that the aerated lagoons are surrounded by conventional stabilization ponds with the area ratio between aerated and conventional cells being in the order of 1:25. Regular spraying by the Mosquito Abatement Department was carried out.

Grease and Scum

Build-up of grease and scum was noted to occur in the corners of the Air-Aqua primary cell, with very little floating material noticed on the surface aerator and Air Gun cells. The Air-Aqua system is designed with a center dike, one of its purposes being to intercept floating material and prevent it from escaping to the effluent.

Odors

In general it may be said that odors emanating from the three systems were of a minor nature and they certainly did not carry beyond the test area. The only exception was the Air-Aqua primary cell which at times released odors that were quite noticeable on the enclosing banks. They were attributed to the floating grease and scum collecting along the banks and in the corners of this cell, where this material would decompose, releasing odors. This occurred primarily along the south banks, and may well have been the result of the very flat slope (8:1) incorporated in the design.

Ice

Ice build-up caused the greatest problems with the surface aeration system. The Air-Aqua and Air-Gun systems were notably free of problems due to ice build-up.

The Air-Aqua system did not cover completely with ice for the winter period and showed the peculiar build-up of "stooks" over the air lines as observed in other installations. Ice thickness found on the Air-Aqua secondary cell varied from approximately 6 inches to 48 inches from inlet to outlet, respectively. With water depth of 10 feet this will cut down the retention period considerably during the winter when water temperatures are adverse to the promotion of biological activity. The foregoing is of course true for all lagoon type systems, be they conventional or aerated. No ice thickness surveys were carried out on the remainder of the test cells due to thin ice.

The Air-Gun system retained open water longer than the Air-Aqua. With the onset of colder temperatures the openings immediately above the guns decreased in size from inlet to outlet. Under severe temperature conditions the openings became covered with ice domes towards the outlet end. Open water conditions existed year-round above the first two or three rows of guns.

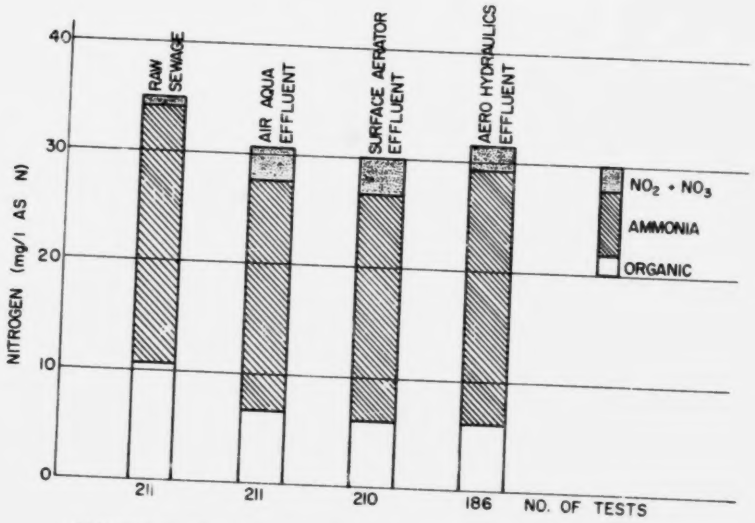

FIGURE 9 Test aerated lagoons average test results nitrogen

Ice built up on the surface aerators to such an extent that it was impossible to clear the ice from the aerator area. The result was that during the winter only three or four surface aerators could be kept operating.

Foaming

As with conventional lagoons or stabilization ponds, foaming conditions were encountered to some degree with all three aeration systems. The surface aeration system showed more foam than the other two systems, due to the more violent agitation of the surface of the water by the aerators.

The quantity of foam generated varied with the water temperature, the maximum condition occurring after spring break-up when water temperatures were rising. However, the foaming did not reach a point where it became a problem

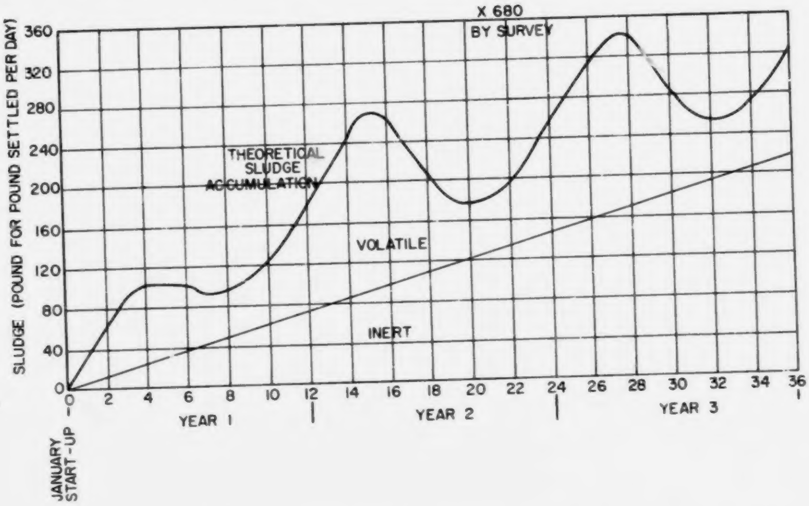

FIGURE 10 Theoretical sludge accumulation in aerated lagoon (Complete anaerobic digestion assumed)

Grit and Rags

After one year of operation considerable quantities of grit and rags were present in all three systems. For the effective long-term operation of any aerated system pre-treatment facilities will be required to remove grit and rags.

Bank Slopes

The following observations on bank slopes were made:

The existing 8:1 slope utilized in the Air-Aqua cells created a wide unaerated band around a portion of the lagoon; this likely resulted in some short circuiting. If flat slopes are required for stability, consideration should be given to providing aeration along the slope. The majority of the dikes for the demonstration lagoons were constructed at 3:1; this slope was stable, but for long-term operation 4:1 is recommended. The slopes of the Air-Gun cell were constructed at 2.5:1 to minimize surface area. These slopes proved unstable under draw-down conditions. The rip-rap provided effective erosion control in all cells.

Equipment Operations

Air-Aqua System. After start-up, difficulties were encountered with pressure build-up in the system. Recommended HCl acid gas cleaning of the tube system was performed without alleviating the problem. Removal of a 250-foot length of tubing revealed that the perforations had not been

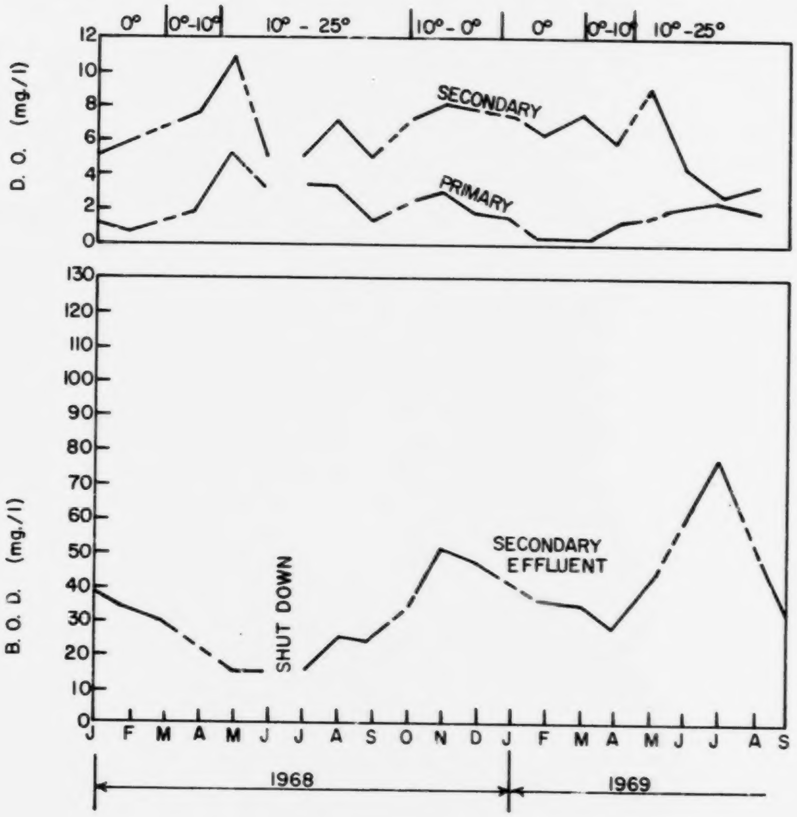

FIGURE 11 Charleswood lagoons monthly trends Air Aqua system

properly cut. All tubing was repunched *in situ*, requiring the lowering of the system so that the workers could walk through the cells feeding the tubing through a punching mechanism mounted on a boat. The repunching took 2 man-days to complete.

Upon completion of this work, pressure on the system dropped to 5.6 to 6 psig. The manufacturer then recommended that acid cleaning be carried out quarterly, regardless of pressure build-up.

When operating properly, the pattern of air distribution should show a grid of air bubbles, released by the perforated tubing. Although this grid did show immediately after installation and repunching of the tubing, it deteriorated after a period of operation due to clogging of the perforations, regardless of acid cleaning, carried out regularly. This clogging had affected the air distribution pattern and was confirmed by D.O. tests. Further investigations traced the loss of pattern to water in the tubes and condensation, and from water coming back through the valves during power failures. By manipulating the acid valves it was possible to force the majority of the water out of the tubes, restoring a reasonably good air pattern.

Supporting equipment such as air-compressors, effluent recirculation pump, flow meters, and samplers were subjected to a regular preventative maintenance program, and few problems were encountered.

Surface Aeration System. Ice has been a problem with the operation of the surface aerators. The basic problem was ice build-up on the impeller shaft and supporting structure, with subsequent freeze-up and stopping of the mechanism. Ice formed on the piles and impeller, resulting in a limited clearance between the rotating ice on the impeller and the fixed ice on the piles. This is a problem that is difficult to overcome with the winter climate experienced in the Winnipeg area. Two methods aimed at overcoming this problem were attempted; firstly, one of the platforms was shrouded with plywood, and secondly a second unit was shrouded with flexible nylon cloth. Both attempts failed.

Other results of the ice accumulation were off-balance, causing vibration of the supporting structure and misalignment of motor reducer, resulting in a coupling failure and loosening of impeller, bending and loss of blades.

In the spring of 1968 the manufacturer replaced all blades with a heavier design. Although this was thought to cure the problem encountered with the impeller blades, subsequent winter operation disproved this, as bending and loss of blades occurred.

It may be possible to reduce or eliminate the icing problems with a change in the supporting structure. The existing structure with four piles provided a surface for the ice to grow on, a two pile arrangement with the piles widely spaced may be more successful.

Air-Gun System. No problems were encountered with the operation of this system until October 1968, when it was noticed that the most northeasterly gun was discharging air continuously. During the winter 1968-69 this condition spread to most guns. It was thought to be caused by the build-up of either ice or rags in the syphon chamber. The latter proved to be true. In May 1969, the manufacturer installed syphon chambers of a new design on 42 guns. The new design, having larger clearances, may eliminate the problems of plugging of the syphon chambers with rags. Insufficient time has elapsed since the modification to allow an assessment of their long-term dependability.

COSTS

Capital Costs

Two sets of cost data have been prepared for aerated lagoons; firstly, the 0.5 Imgd demonstration lagoons and secondly, a general unit capacity cost curve.

All costs and estimates in this report are adjusted to an Engineering News Record Sewage Treatment Plant cost index of 132.0. As costs change the appropriate adjustment must be made to the reported costs.

The capital costs for the demonstration lagoons are shown in Table 2. These costs have been arrived at by assuming each system would be as indicated in Figure 1, insofar as floor dimensions and identical equipment layout are concerned. However, in order to relate realistic costs to a permanent installation it is necessary to assume each unit is to be constructed as a totally independent unit. An independent unit is one with independent piping, diking, influent-effluent structures and electrical supply. Therefore, only items 1 and 2 of Table 2 are established directly from the contract amounts. The remaining items were arrived at by combining estimated quantities and the actual unit prices tendered. The general contractor for the lagoon project was consulted on the tendered prices. All costs were considered realistic with the exception of rip-rap. In the case of rip-rap, the contractor was of the opinion that the price should be 1.5 times the tendered price (i.e. $12.00/cu. yd. in place).

The estimated quantities for earthwork were calculated on the basis of 3 feet freeboard, 12 feet roadway width, and 4:1 dike slopes, rip-rap facing the interior dikes from toe of slope to 1 foot above the normal operating level. These adjustments are considered necessary to insure a permanent and relatively maintenance-free structure with the soils encountered in the Winnipeg area.

Sodding and seeding quantities include sodding the interior dikes above the rip-rap and 50% of the total dike crest, and seeding the exterior dike slopes. Roadway quantities are based on an asphaltic surface treatment being applied to the unsodded dike crest.

The electrical costs include service entrance equipment, motor starters, and lighting for the equipment and structures associated with the respective systems. Power distribution costs were based on the billing received from Manitoba Hydro for the supply and erection of equipment, and allocation to the respective systems on the basis of length of cable and number of poles required for each.

Actual influent and effluent chamber and piping costs were reestimated on the basis of piping and chambers being of such length and size for an independent treatment system of 0.5 Imgd capacity. Fencing costs are for the perimeter of the cell taken at the toe of the dike slope. Additional items of chlorination facilities and instrumentation are included since these would be desirable in a permanent installation. The costs do not include pumping station, forcemain, outfall, or land. However, they are inclusive of engineering, legal and administration charges.

From Table 2 it can be seen that the total costs for the Air-Gun are slightly lower than the Air-Aqua and Surface Aerator Systems. Major cost differences are due to additional rip-rap and

TABLE 2

Capital Costs - Demonstration Lagoons

	Air-Aqua	Surface Aerator	Air-Gun
1. Aeration Equipment, Supply (Blowers, Headers)	$ 47,500	$ 44,300	$ 38,000[1]
2. Aeration Equipment, Installation (Headers, Housing, Platforms)	23,000	56,500	33,700[2]
3. Electrical	3,500	7,800	3,500
4. Influent Piping	3,200	3,200	3,200
5. Effluent Piping	3,500	2,200	2,200
6. Influent Chamber	17,900	17,900	17,900
7. Effluent Chamber	22,900	22,900	22 900
8. Excavation & Dike Construction	32,400	29,200	27,000
9. Clearing, Grubbing Unsuitable Material	10,800	10,800	10,800
10. Rip-rap	93,000	73,500	70,300
11. Seeding & Sodding	6,400	5,000	4,000
12. Roadway (asphalt)	3,900	3,500	3,200
13. Chlorination Facilities (including housing)	20,000	20,000	20,000
14. Power (hydro)	3,500	5,400	3,500
15. Instrumentation (including magnetic flow meter)	10,000	10,000	10,000
16. Fencing	4,700	4,000	3,500
Total Estimated Cost	$306,200	$316,200	$273,700
10% Eng. & Contingencies	30,600	31,600	27,300
Total Estimated Cost	$336,800	$347,800	$301,000

Costs indexed to ENR S.T.P.
cost index 132.0

1) Includes installation, excludes blowers and header
2) Includes blower and header

excavation required for the dividing dike, greater dike perimeter for the Air-Aqua System, and platform costs for the surface aerator.

Figure 12 shows the development cost of the aerated lagoon process per unit of lagoon capacity. These costs have been developed by applying the unit material and installation costs of the demonstration lagoons to calculated quantities. Equipment supply and installation costs were derived from equipment manufacturers' quotations for similar facilities at other treatment works.

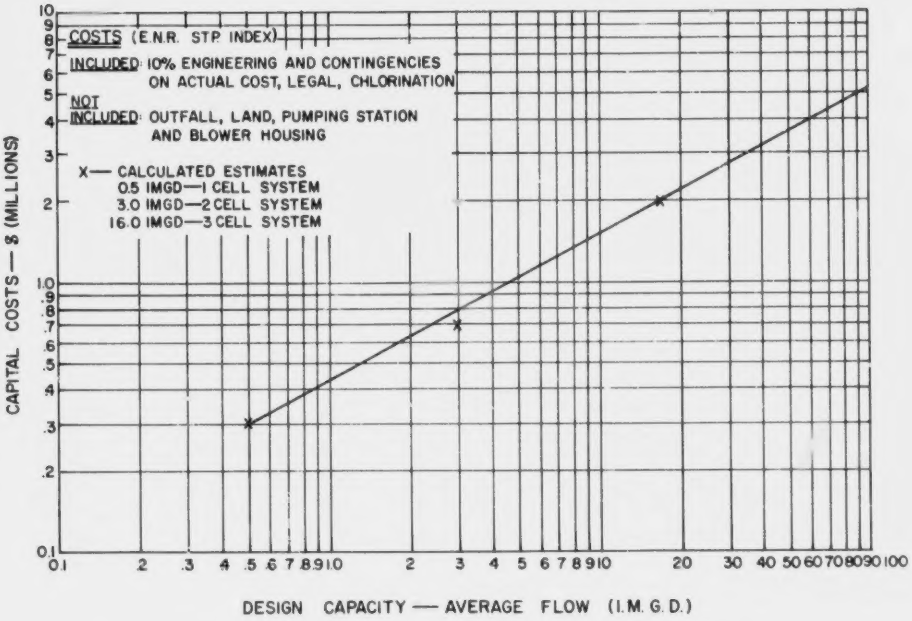

FIGURE 12 Aerated lagoons capital cost vs. design capacity

Operation and Maintenance Costs

The operating and maintenance costs for the 0.5 Imgd demonstration lagoons are shown on Table 3. They are based on 17 months of operation.

Assuming chlorination of the effluent at 8 mg/l is desirable, the total operating costs should be adjusted to $53, $107 and $56 for the respective lagoons.

The power costs tabulated in Table 3 are based on field measurements of true power drawn with a unit power cost of 0.9c/kwh. Allocated to each system is a shore of the heating and sampling pump load. In the case of surface aeration the cost is calculated on all eight units operating. In terms of oxygen requirements, the surface aerators are capable of treating in excess of the rated flow. However, at the time of design, eight units were considered necessary to insure adequate mixing.

Repairs include regular weekly inspection and servicing of equipment, non-routine repairs and replacement parts where required. The costs of repunching the aeration tubes and replacing the siphons on the Air-Guns are not included in Table 3. The labor and materials on the servicing were based on actual time card and material receipts and allocated to the respective systems. Labor includes a 25% factor for overhead. General costs include maintenance to the common inlet chamber, metering equipment, etc. Since it was difficult to allocate these costs to a specific lagoon they were split equally to the three systems. Laboratory costs include labor and supplies for

TABLE 3

Actual Operating and Maintenance Costs
0.5 Imgd Demonstration Lagoons

	Avg. Cost per Img Treated		
	Air-Aqua	Surface Aerator	Air-Gun
Power	$ 10	$ 64	$ 15
Equipment Servicing & Repair	13	14*	12
Laboratory & Control	13	13	13
Road & Dike Maintenance	4	3	3
Snow Removal, Grass Cutting, Mosquito Control	3	3	3
General	3	3	3
Total Operating & Maintenance Costs per Imgd Treated	$ 46	$100	$ 49

*Not continuously maintained due to ice.

sampling and analysis. Actual laboratory costs were three times the cost reported due to the frequency and number of analyses performed under test conditions. Therefore, the lower costing for laboratory reported here would be a more realistic value under normal operating conditions. Maintenance and laboratory personnel were not based at the lagoons during the test; therefore considerable labor and vehicle time for travelling were not charged to the test lagoons.

The development of operating and maintenance costs for aerated lagoons is shown in Figure 13. The curve was prepared by projecting the actual demonstration lagoon costs on the trend line as indicated. The slope of the line was guided by costs reported in recent literature (Okey and Rickles, 1968). The reported costs from the literature reference have been adjusted to Imperial Gallons and are also shown on Figure 13. It is noteworthy that the referenced costs are representative of power costs of 1 c per kw. hr. and labor of $5.00/hr., while in comparison the local power and labor costs are approximately 0.9 c per kw. hr., and $4.25 per hr., respectively. Very little cost information at other aerated lagoons is available to confirm the trend costs for increasing capacities. However, a check was made on the expected operating and maintenance costs by determining the future power requirement for mechanical equipment coupled with current power rates. The total costs were extrapolated from the power costs on the basis of power being about one-third of the total costs. This provided a reasonable check on the trend developed by the literature.

CONCLUSIONS

Aerated lagoons were found to be capable of providing "secondary equivalent" sewage treatment. Under prairie climatic conditions there is a problem of sludge accumulation leading to a decline in efficiency of BOD removal and a reduction in the dissolved oxygen concentration during the summer months.

The economic feasibility of aerated lagoons is questionable until the extent and cost implications of the sludge problem are fully defined by further research and experience.

The use of surface aerators under prairie winter conditions is not practical due to ice build-up

It can be concluded that aerated lagoons are an effective means of providing secondary treatment but some provision must be made for sludge handling.

It is intended to continue the investigation on a less rigorous scale, with a goal of determining the long term effects, and finding a practical and economical solution. Equipment manufacturers are actively pursuing a solution.

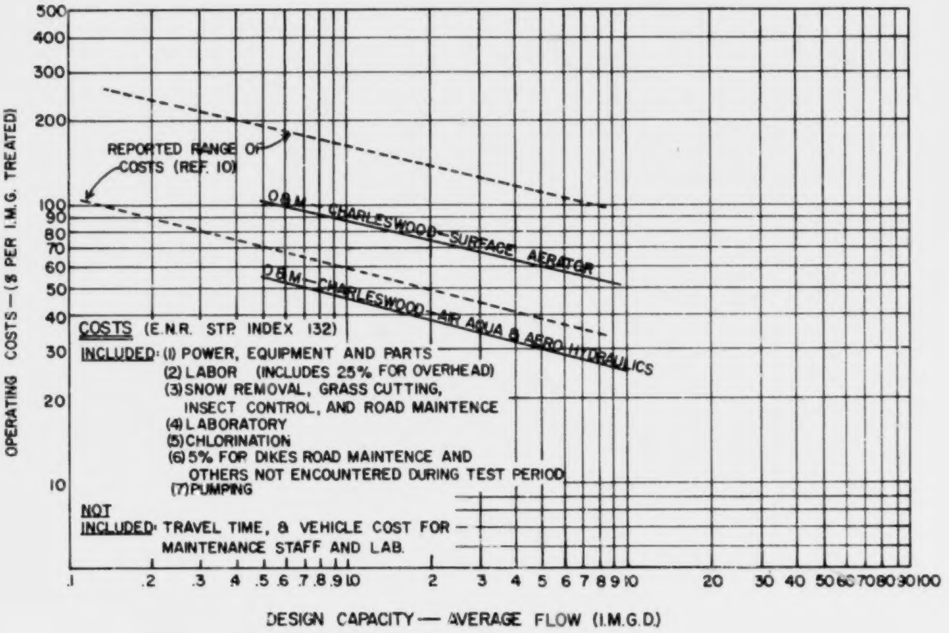

FIGURE 13 Aerated lagoons operating and maintenance costs

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DESIGN CONSIDERATIONS FOR EXTENDED AERATION IN ALASKA

Sidney E. Clark, Harold J. Coutts and Conrad Christianson

INTRODUCTION

Alaska is the largest state of the United States, sparsely populated and with a variety of climates, including arctic, sub-arctic, marine sub-arctic and temperate. The population is small and widespread with 294,417 people (preliminary 1970 census figure) inhabiting 586,000 square miles of land area. Settled areas requiring domestic sewage treatment include large municipalities, military installations, remote sites and villages, each of these having different requirements and presenting different problems.

Construction and power costs in Alaska are very high in general, and excessively so in remote areas (1.5 to 5 times Seattle construction costs index). Skilled personnel for operation of treatment plants are scarce and, in most cases, nonexistent.

The effect of man's waste on arctic and sub-arctic ecosystems has received little attention in the past and is not well understood. Because of recent increased interest in the Arctic region, some information is now becoming available on man's possible influence. For example, during the winter, dissolved oxygen (DO) of ice-covered Alaska rivers may reach extremely low levels of 3 mg/l or less under natural conditions (Frey, 1969; Gordon, 1970; Roguski, personal communication). Because of the retarded ability of Alaska streams to replenish DO under total or nearly total ice cover, it becomes essential that the natural balance not be upset by man. Under these conditions, secondary sewage treatment will be required to assure adequate stream protection.

One of the major advantages of biological processes for provision of secondary treatment is their ability to oxidize waste without large inputs of energy, thus reducing shipping costs, etc., associated with materials required for chemical treatment. All factors considered, extended aeration systems have considerable potential for reliable and economical secondary treatment at larger governmental installations and large communities in Alaska (populations greater than 250).

Current extended aeration research is being conducted by several groups:

1. Corps of Engineers
   Cold Regions Research and Engineering Laboratory
   Alaska District

2. University of Alaska
   Institute of Water Resources

3. Federal Water Quality Administration, Alaska Water Laboratory
   Cold Climate Research Program

Waste treatment research at the Alaska Water Laboratory is concerned primarily with adapting methods developed in the contiguous United States to the extreme cold climates found in Alaska. The scope of the present work on activated sludge is, in general, limited to extended aeration, and includes investigations in the following areas:

1. Low temperature biokinetics
2. Low temperature solids removal
3. Degree of environmental protection required for equipment and processes.
4. Aeration requirements
5. Aeration chamber mixing
6. Waste sludge characteristics and disposal

The above investigations are being conducted on a laboratory and pilot plant scale. Monitoring of existing facilities is also taking place.

LITERATURE AND EXPERIENCE REVIEW

Low Temperature Biological Treatment Feasibility

Although the activated sludge process is affected by temperature, operation at temperatures approaching freezing is feasible. A number of investigators have reported a considerable amount of biological activity taking place at freezing temperatures and below (Ayres, 1962; Halvorson, 1962). Miller (1967) has reviewed the information available on microorganisms indigenous to cold environments and found that research on psychrophilic organisms is still in the initial stage, but concluded that "truly psychrophilic microorganisms do exist and are distinguished by their ability to grow at very low temperatures and to do so at rates comparable to those of mesophiles at higher temperatures." The feasibility of effective biological treatment by full-scale extended aeration facilities at operating temperatures as low as 2° C has been demonstrated (Anonymous, 1965; Grube and Murphy, 1968; Schmidtke, 1967).

Temperature Effects

Pasveer (1955) conducted laboratory scale temperature studies with activated sludge and reported that the process goes on almost as well at 3° - 5° C as it does at 20° C. Wuhrmann (1956) found in his studies of the activated sludge process that "the BOD removal seems to be only slightly influenced by temperature, whereas nitrification is markedly higher in summer than in winter." Ludzack (1965) conducted bench scale studies using a continuous apparatus with a detention time of 24 hours and a loading of 35 lb. COD/1,000 ft^3 and demonstrated COD removal efficiencies of <90% at 21° - 25° C, 90% at 10° C and 84% at 5° C. Hunter et al. (1966) conducted batch operated laboratory scale studies on the effect of temperature and retention times on the activated sludge process. At temperatures between 4° C and 45° C, they found little change of BOD or suspended solids removal efficiencies. As the temperatures increased, they found less filamentous growth and increased protozoa and rotifer populations. Grube and Murphy (1968) evaluated an oxidation ditch and found BOD removal efficiencies greater than 90% with liquid temperatures of 2° C, air temperatures ranging down to -40° C, and average detention times of 2.3 days. Influent temperatures averaged 16.6° C with a minimum of 7.5° C. Gustaffson and Westbury (1965) evaluated the activated sludge process for application at Kiruna, Sweden, and obtained 75% BOD reduction with a 3½ hour detention time system at 2.8° - 4.8° C and 2,700-3,500 mg/l MLSS.

Temperature Coefficient

The temperature coefficient, θ, is used in the relationship

$$k_1/k_2 = \theta\ (t_1 - t_2)$$

to define the effect of temperature on biological activity. The values k_1 and k_2 refer to velocity constants at temperatures t_1 and t_2 respectively. The value of θ indicates the extent of the temperature effect on the biological activity. Use of this equation, known as the Arrhenius relationship, to define the effect of temperature on wastewater and reaction rates, dates back to Streeter and Phelps (1925) and Theriault (1927), who reported θ values of 1.047 for domestic wastewater and river water (Zanoi, 1969). Pohl (1967) concluded that θ was dependent on the mixed liquor concentrations: $\theta = 1.038$ at low MLSS and 1.000 at high MLSS. Benedict (1968) conducted studies in the temperature range of 4° - 32° C and concluded θ ($\theta = 1.078$ @ 4° C) was independent of loading when the loading rate did not exceed 0.53 lbs BOD/day/lb/MLSS, but θ increased as loadings above 0.53 were imposed. Eckenfelder (1967) suggested that θ, based on overall treatment efficiencies, was a function of the organic loading and reported θ values for activated sludge of 1.00 at low loadings and 1.02 at high loadings.

Solids Separation

Solids removal plays a very important part in the efficiency of the activated sludge treatment process. The degree of sludge separation directly influences the quality of effluent from wastewater treatment plants with higher concentrations of effluent solids contributing to high effluent BOD. Reed and Murphy (1969) conducted an investigation on settling characteristics of activated sludge at temperatures ranging from 1.1° to 23.4° C and found that the influence of temperature on settling velocity decreased as the concentration increased. They developed an equation for zone settling based on experimental data. They also suggested upflow sludge blanket clarifiers as having greater potential for cold regions application. Benedict (1968) suggested that the effect of sludge settleability on gross COD removal was magnified at low temperatures and as the loading rate was increased.

Hansen (1967) reported on a method of solids separation which successfully employed shallow depth sedimentation theory. The settling units consisted of small diameter tubes (1-inch) inclined at 5° and 2-4 feet in length. Detention times were very short and backwashing was necessary for removal of accumulated solids. Hansen (1967) also reported on the use of steeply inclined tubes (60°) which permit solids deposited in the tubes to continuously slide down by gravity. A secondary clarifier of a trickling filter plant was converted to a biological reactor and the steeply inclined tubes utilized for solids separation which increased plant efficiency from 85% to more than 95%. The effluent suspended solids averaged 70 mg/l varying from a low of 7 mg/l to a high of 190 mg/l, which was comparable to that produced by a conventional clarifier of an extended aeration plant of the same capacity (3000 gallons per day at 12-hour detention). Other reports are available which describe the use of tube settlers in water treatment and waste treatment solids separation (Culp, 1969; Schlecta and Hsiung, 1969).

Pohl (1970) investigated tube settlers in the laboratory and obtained the best results at room temperature but found the tubes passing excessive collodial solids occasionally.

Design Parameters

Little information is available on biological treatment process design for temperatures less than 5° C. Ludzack (1965) and Hunter, et al. (1966) observed that excess MLSS accumulation increased with decreasing temperature. The cell yield (c) increases with increasing temperature because it is believed a larger portion of BOD removed is utilized for energy at low temperatures than at high temperatures (Stewart, 1964). Since the rate of endogenous respiration is depressed at low temperatures, the quantity of excess sludge produced is increased. Benedict (1968) reports values for c and k (endogenous rate) at 4° C of 0.42 mg/mg COD_r and 1.32% respectively.

Aeration

Eckenfelder and O'Connor (1961) stated that the temperature coefficient θ, when applied to oxygen transfer efficiencies, has been reported to vary from 1.016 to 1.047 and that studies on bubble aeration indicated a temperature coefficient of 1.02 applied. The effects of temperature on stream reaeration have been studied under controlled experiments in the laboratory (Anonymous, 1961). A value for θ of 1.0241 for the temperature range of 5° to 30° C was found. Black (1968) described a procedure for evaluation of aeration devices and stated that a θ value of 1.030 or higher should be used for cold water.

LABORATORY STUDIES

During the past two years, three bench scale activated sludge reactors have been utilized for kinetics and solids separation studies. The three units are illustrated in Figures 1, 2, and 3. The systems have been operated as continuous flow systems with the feed being primary effluent

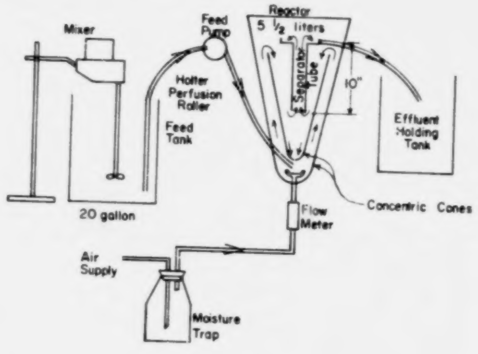


FIGURE 1 Cone reactors (as manufactured by Pope Scientific)

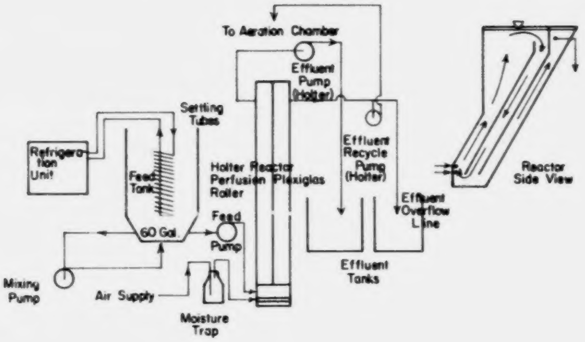

To Aeration Chamber
Effluent Pump (Holter)
Settling Tubes
Refrigeration Unit
Feed Tank
Holter Reactor Perfusion Roller
Plexiglas
Effluent Recycle Pump (Holter)
Reactor Side View
Effluent Overflow Line
Feed Pump
60 Gal.
Effluent Tanks
Mixing Pump
Air Supply
Moisture Trap

FIGURE 2 AWL reactor (8.9 gal.)

brought to the laboratory from the Eielson primary sewage treatment plant. Routine analysis included influent and effluent BOD and COD, mixed liquor and effluent suspended solids (SS) and volatile suspended solids (VSS). Nutrient analyses of the influent and effluent samples were made weekly and included ammonia, nitrite, nitrate, organic nitrogens, total phosphates, and orthophosphates. A limited number of coliform counts were made on the influent and effluent. Microscopic examinations of the reactor contents were made on an irregular basis at times when apparent or suspected changes in the mixed liquor had taken place. The examinations consisted of general observations on the relative quantities of protozoa present and the degree of activity. BOD, COD, and solids analyses were done in accordance with **Standard Methods** procedures (1965). Coliform counts were made by the membrane filter method as described in **Standard Methods** and nutrient analyses were made in accordance with Federal Water Quality Administrations Standards (1969).

The cone reactors (Fig. 1), when operated at 1.3° C and 6.5° C for long periods of time, showed some interesting characteristics which are summarized in Tables 1 and 2. Both biological sludges were relatively easy to establish.

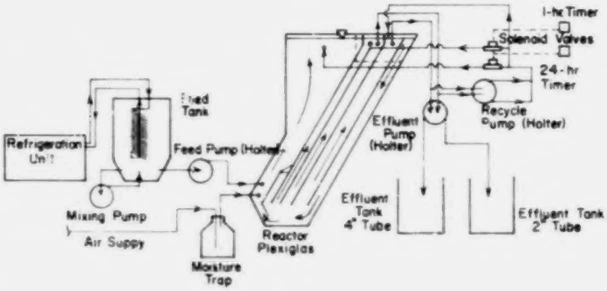

1-hr Timer
Solenoid Valves
24-hr Timer
Feed Tank
Recycle Pump (Holter)
Effluent Pump (Holter)
Refrigeration Unit
Feed Pump (Holter)
Mixing Pump
Effluent Tank 4" Tube
Effluent Tank 2" Tube
Air Suppy
Reactor Plexiglas
Moisture Trap

FIGURE 3 AWL reactor (12.45 gal.)

The reactor runs started with the longest detention time first and the times decreased in chronological order. The 1.3° C reactor took a considerable amount of time to establish a stable system (more than 3 months). However, a good removal rate was obtained before the MLSS stabilized. There was apparently little difference in the biological activity at the two temperatures, but operation of the reactor at 6.5° C was more erratic.

Both reactors generally showed "auto-induced sludge wasting" in the same manner as the College Utilities oxidation ditch described by Grube and Murphy (1968). The MLSS would build up to a point and begin to pass solids for 1 or 2 days and then repeat the cycle. The cycle was repeated within 2 to 3 weeks as opposed to the monthly occurrence reported by Grube and Murphy.

The reactors differed in their manner of passing solids, with the 1.3° C reactor generally having a much more turbid effluent and the 6.5° C reactor having a relatively clear effluent. Heavy solids passed from the 6.5° C reactor by rising in the settling tube as a solid mass. As the concentrations of solids in the mixed liquor increased, the level of solids in the settling tube would rise until spilling over into the effluent tank. After passing an undetermined amount of solids, the cycle would be repeated. A gradual drop in pH was noted in the 6.5° C unit as the suspended solids began to build before discharging. The pH dropped from slightly above 7 to values of 6.6 to 6.7. pH of the 1.3° C unit consistently remained around 7.4. The 6.5° C effluent solids settled to the bottom of the effluent tank leaving a clear liquid above, whereas, the 1.3° C effluent solids did not settle to any degree. As the 1.3° C reactor became more stabilized, the effluent became less turbid

TABLE 1

Data Summary
1.3° C Cone Reactor
Feed: Primary Plant Effluent

Detention Time (hrs)	21	15	13	9
Influent BOD (mg/l)	111	170	201	184
Reactor				
Susp. Solids (mg/l)	1,074	1,561	2,657	2,926
Volatile Susp. Solids (mg/l)	890	1,324	2,212	2,402
Filtered Effluent				
BOD (mg/l)	37	11	20	14
% BOD Removal	66	93	90	92
Unfiltered Effluent				
Susp. Solids (mg/l)	29	43	38	82
BOD (mg/l)	40	62	28	44
% BOD Removal	64	64	86	76
Loading Factor $\frac{\text{lb BOD Fed}}{\text{lb MLVSS-Day}}$	.19	.21	.17	.20
Product of MLVSS and Det. Time	14,700	19,500	28,000	21,600

and the MLSS began to increase. The 6.5° C reactor operation was less stable, with the maximum level of MLSS generally not rising above 2,300 as opposed to 3,000 for the 1.3° C MLSS. Results of nutrient analyses are presented in Table 3. There was a significant change in nitrate and total nitrogen at 6.5° C when going from 9 to 13 hours detention time. This was also true at 1.3° C to a lesser degree. There was a greater reduction in ammonia nitrogen and a greater increase in nitrate nitrogen at 6.5° C. Ammonia was essentially not affected at 1.3° C. Total nitrogen removals were much higher at 6.5° C than at 1.3° C with little detention time effects.

Overall results of operation of the 8.9 gal. and 12.45 gal. reactors are presented in Tables 4 and 5. Temperature changes were accomplished by a gradual increase or decrease in the constant temperature room temperature. These reactors were operated at 12-hour hydraulic detention times with daily sludge wasting to maintain the MLSS at 4,000 mg/l. The 8.9 gal. reactor was later converted to a 24-hour operation. Sludge was wasted by drawing off the required amount of mixed liquor. A portion was used for a solids analysis to determine the exact amount of solids removed. The effluent BOD and COD figures of 9 to 21 mg/l and 46 to 96 mg/l indicate that a considerable amount of biological activity takes place at low operating temperatures.

Effluent BOD/COD ratios varied from 0.13 to 0.27, indicating that effluent organics were well oxidized. These were in comparison with the influent BOD/COD ratios of 0.55 to 0.66.

TABLE 2

Data Summary
6.5° C Cone Reactor
Feed: Primary Plant Effluent

Detention Time (hrs)	17	15	13	9
Influent BOD (mg/l)	139	132	153	155
Reactor				
Susp. Solids (mg/l)	2,346	1,885	1,880	2,285
Volatile Susp. Solids (mg/l)	1,915	1,563	1,587	1,801
Filtered Effluent				
BOD (mg/l)	51.3	16.3	13.3	11.7
% BOD Removal	63	88	91	92
Unfiltered Effluent				
Susp. Solids (mg/l)	11	69	96	45
BOD (mg/l)	53	36	31	33
% BOD Removal	62	73	80	79
Loading Factor $\frac{\text{lb. BOD Fed}}{\text{lb. MLVSS-Day}}$	.08	.106	.18	.23
product of MLVSS and Det. Time	31,600	23,000	23,300	20,600

TABLE 3

Cone Reactors Results of Nutrient Analysis

13 Hour Detention Time

	1.3°C REACTOR			6.5°C REACTOR		
	Influent	Filtered Effluent	Unfiltered Effluent	Influent	Filtered Effluent	Unfiltered Effluent
NH_3-N (Ammonia)	22	19	18	19	1	1
NO_2-N (Nitrite)	.13	.09	.05	.11	.13	.15
NO_3-N (Nitrate)	.13	2.13	2.02	.21	9.17	12.13
Kjeldahl-N (Nitrogen)	41	28	29	37	3	3
Total (Nitrogen)	41.26	30.22	31.07	37.32	12.30	15.28
Total Nitrogen Removals (%)	--	27	25	--	67	59
O-PO_4 (Ortho-Phosphate)	20	18	18	19	18	18

9 Hour Detention Time

	1.3°C REACTOR			6.5°C REACTOR		
	Influent	Filtered Effluent	Unfiltered Effluent	Influent	Filtered Effluent	Unfiltered Effluent
NH_3-N (Ammonia)	21	19	19	21	1	1
NO_2-N (Nitrite)	.03	.03	.03	.06	.14	.12
NO_3-n (Nitrate)	.11	.68	.54	.07	8.03	14.45
Kjeldahl-N (Nitrogen)	36	26	27	35	3	3
Total (Nitrogen)	36.17	26.71	27.57	35.13	11.17	17.57
Total Nitrogen Removals (%)	--	26	24	--	68	50
O-PO_4 (Ortho-Phosphate)	17	14	15	19	18	16

(1) Total nitrogen results reported are the sum of the nitrite and Kjeldahl nitrogen analysis.

The amounts of sludge wasted varied from 0.42 mg susp. solids/mg BOD removed at the low temperatures to 0.14° to 10.5° C and 24-hour detention time. The pH of both reactors ranged from 7.2 to 7.6 during the sample periods reported.

Poor settling sludges were developed during operation of these reactors with the Sludge Volume Index (SVI) consistently ranging above 200. The sludge produced appeared to be of a zoogloeal type similar to that reported by Heukelekian and Wiesburg (1965) who found a direct correlation between increasing SVI and increasing bound water for this type of bulking. Very little evidence of *Sphaerotilus* was noted during microscopic examination. Ludzack (1965) also reported a poor settling sludge at low temperatures (5° C) with very poor drainability.

The significance of protozoa in an efficiently operating activated sludge process, as reported by McKinney (1956), was observed during operation of the reactors even at the coldest temperatures. The 12.45 gal. reactor was seeded at temperatures <2° C with return sludge from an oxidation ditch treating domestic sewage. Initially, the effluent was very turbid as the sludge was acclimating

TABLE 4

Summary of Results of 8.9 Gallon Reactor and 12.45 Gallon Reactor at 12-hr Detention

Feed: Primary Plant Effluent

	REACTOR TEMPERATURE (AVG°C) .6	2.9	3.8	8.0
Reactor MLSS (mg/l)	4160	4097	4076	3737
% VSS	80	80	81	80
BOD	2489	2503	1477	1299
COD	5648	5788	5260	4705
Loading: $\frac{\text{lb. Infl. BOD}}{\text{lb MLVSS-Day}}$	0.12	0.10	0.10	0.14
Sludge Wasted: $\frac{\text{mg/MLSS}}{\text{mg BOD Removed}}$	0.42	0.33	0.32	0.33
Unfiltered Effluent				
Suspended Solids (mg/l)	18	3	12	5
BOD (mg/l)	21	13	17	9
BOD Removal (%)	89	92	90	96
COD (mg/l)	78	46	67	96
COD Removal (%)	76	73	78	83
BOD/COD Ratio				
Influent	0.60	0.60	0.55	0.66
Effluent	0.24	0.23	0.25	0.16
Reactor	0.44	0.43	0.28	0.28

itself to the new conditions. The decreasing turbidity of the sludge as acclimation progressed corresponded to increasing numbers of protozoa, generally *Paramecium* and *Vorticella*. As reported by McKinney (1956), a very well-stabilized activated sludge system will have few stalked ciliates and no other protozoa because of relatively few bacteria, whereas, a somewhat less stabilized system will have greater numbers of free-swimming ciliates because of greater numbers of free-swimming bacteria. He stated that the presence of stalked ciliates indicates an activated sludge system with a low BOD effluent. *Vorticella* was present in both reactors after initial startup except for one period in the 12.45 gal. reactor as described below.

After stable operation at temperatures $<2^\circ$ C and $\sim 4^\circ$ C the 12.45 gal. reactor temperature was increased to 8° C over a period of 6 days. The effluent suspended solids increased from approximately 5 mg/l before the temperature increase to approximately 18 mg/l during the increase and reached a maximum of 46 mg/l after 3 days at 8° C. During this period, the effluent became turbid with few solids settling out in the effluent tank. The protozoa became very reduced in numbers and inactive. Again, the return to normal operation corresponded to an increase in the number of *Vorticella* and *Paramecium* present in the sludge. Coliform removal also corresponded directly to the numbers of protozoa present, dropping from 99.8% removal before the upset to less than 80% during the protozoa number reduction. Ten days after returning to stable operation at 8° C, the sludge was exhibiting the same characteristics as with the 8.9 gal. reactor. That is, the

SVI was ranging around 250 and the floc exhibited a fluffy snowflake appearance. Operation of the reactor was not impaired under these conditions because a backwash cycle was added to the settling apparatus. Protozoa increased in numbers when the systems stabilized at 12° C.

Sludge wasting and disposal in cold climates should be given attention. Based on data presented earlier, it would appear that provision should be made for wasting 0.5 lb. solids per lb. or BOD removed at colder operating temperatures (<5° C) and at organic loadings of 0.1 lb. influent BOD per lb. MLVSS-Day. Sludge digestion and disposal methods present a problem at colder temperatures due to added heat requirements and poor drainability. Ludzack (1965) indicated that sludge development at cold temperatures may require digestion at higher temperatures before disposal. Thomas (1950) indicated the freeze-thaw cycle may be taken advantage of in cold climates to increase drainability.

Tube settlers have been evaluated as a possible alternate means of providing solids separation and return. During operation, sludge rises in the tubes until it reaches a level at which it is in equilibrium with the effluent flow. Action in the tube consists of a rolling motion in which solids are being carried up along the top side of the tube in a mass with the effluent, as shown in Figure 4. The mass gradually settles toward the bottom side of the tube where it enters a current moving downward caused by the weight of the solids. During normal operation, solids in the tube are constantly being replaced at a relatively high rate. In the temperature range of 0° through 4° C the SVI of the mixed liquor ranged around 230 and did not hinder the operation of the reactor. At 8°

TABLE 5

Summary of Results of 8.9 Gallon Reactor at 24-hr Detention

Feed: Primary Plant Effluent

	REACTOR TEMPERATURE (AVG°C)		
	1.9	6.8	10.5
Reactor MLSS (mg/l)	2595	3872	3896
% VSS	83	83	82
BOD	1693	2105	1808
COD	3712	5019	5178
Loading: lb. Infl. BOD / lb MLVSS-Day	0.07	0.07	0.07
Sludge Wasted: mg/MLVSS / mg BOD Removed	0.42	0.16	0.14
Unfiltered Effluent			
Suspended Solids (mg/l)	3	4	6
BOD (mg/l)	14	10	10
BOD Removal (%)	93	95	95
COD (mg/l)	51	53	69
COD Removal (%)	83	84	80
BOD/COD Ratio			
Influent	0.66	0.66	0.62
Effluent	0.27	0.19	0.13
Reactor	0.46	0.42	0.35

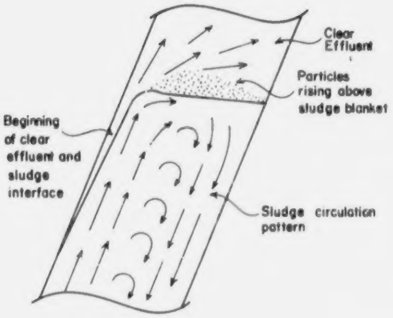

Clear Effluent
Particles rising above sludge blanket
Beginning of clear effluent and sludge interface
Sludge circulation pattern

FIGURE 4 Sludge action in upflow clarifier settling tubes

C and above, the SVI increased to values of 260 and greater and the sludge took on a fluffy snowflake appearance. The rolling action of the sludge in the tubes stopped and the sludge height began to rise, eventually spilling out with the effluent. Cutting the effluent flow rates back to less than 0.2 gpm/ft^2 resulted in lowering the DO in the effluent tubes to zero, which further complicated the problem. The studies indicate that some means for backflushing tube-settler controlled upflow clarifiers must be provided if mixed liquor concentrations greater than 2,000 mg/l are to be achieved with reliable operation.

The 12.45 gal. reactor was operated for a period of time with a very low continuous overflow rate and then increased to an average rate of 0.5 gpm/ft^2 with an alternating on-off cycle. In other words, with the on 1/2 hour - off 1/2 hour cycle, the actual flow was 1 gpm/ft^2 for 1/2 hour. The SVI again ranged above 200 with very consistent solids removal. The effluent solids concentrations were very low for the whole range of studies. The longer on times for the on-off cycle (2-1/2 hours on as opposed to 1/2 hour) did indicate that longer cycles may result in higher effluent solids concentration. Summaries of the results obtained at various temperatures and overflow rates are presented in Tables 6 and 7, and Figures 5 and 6. Adding a backwash cycle provided a definite advantage in that it prevented a bulky sludge from becoming stagnant in the tubes.

Indications are that sludge bulking probably is a general problem in the activated sludge process at colder operating temperatures and special precautions in design will be necessary to assure effective solids control. This problem was reported by Ludzack (1965). Bulking sludges have not been reported in cold temperature oxidation ditch studies (Anonymous, 1965; Grube and Murphy, 1968); however, these ditches were operated at much longer detention times (1.6 to 2.3 days), which may be a factor. Downing (1968) showed that settleability is improved by longer detention time (>10 hrs) and very short detention time (<5 hrs) when operating an activated sludge plant at warm temperatures. At any rate, indications are that backwashing, in conjunction with lower overflow rates, will overcome this problem.

Pilot Plant

In cooperation with the Alaskan Air Command, the Alaska Water Laboratory constructed and operated a pilot waste treatment facility at Eielson Air Force Base (EAFB). The facility included

TABLE 6

8.9 Gallon Reactor Results of Operation with Varying Effluent Overflow Rates on the Settling Tubes

	INFLUENT			REACTOR			EFFLUENT				
Reactor[1] Temp (° C)	Susp. Solids (mg/l)	BOD (mg/l)	COD (mg/l)	Susp. Solids (mg/l)	SVI	Overflow Rate (gpm/ft²)	Susp. Solids (mg/l)	BOD (mg/l)	% BOD Removal	COD (mg/l)	% COD Removal
.35 (.3-.5)	95	244	292	3973	238	.4	10	12	95	69	76
.7 (.4-.9)	112	253	370	4237	238	.3	8	22	91	71	79
						.6	20	29	89	87	77
4.2 (2.8-6.4)	94	193	229	4147	--	.3	10	17	91	60	74
						.6	14	20	90	70	69
3.8 (3.5-4.1)	77	142	283	4067	229	.5	10	14	90	62	78
						.8	13	20	86	69	76

(1) Values in parenthesis are minimum and maximum for that period.

TABLE 7

12.45 Gallon Reactor Results of Operation with Varying Effluent Overflow Rates on the Settling Tubes

	INFLUENT			REACTOR			EFFLUENT					
Reactor[1] Temp (° C)	Susp. Solids (mg/l)	BOD (mg/l)	COD (mg/l)	Susp. Solids (mg/l)	SVI	Overflow[2] Rate (gpm/ft²)	Tube Size	Susp. Solids (mg/l)	BOD (mg/l)	% BOD Removed	COD (mg/l)	% COD Removed
2.4 (1.4-3.5)	77	177	303	3957	--	.2 (continuous)	2 x 3.5	2	19	89	35	88
							4 x 3.5	2	19	89	39	87
2.9	86	185	275	4157	214	.3 (on 1/2 hr off 1/2 hr)	2 x 3.5	4	12	94	50	82
							4 x 3.5	3	10	95	52	81
4.4 (4.0-4.7)	93	223	321	4095	235	.5 (on 1/2 hr off 1/2 hr)	2 x 3.5	4	12	95	55	83
							4 x 3.5	5	12	95	69	79
7.8 (6.8-8.4)	87	194	313	4504	209	.5 (on 1/2 hr off 1/2 hr)	2 x 3.5	12	20	90	69	78
							4 x 3.5	14	23	88	64	80

(1) Values in parenthesis are minimum and maximum for that period

(2) Notes in parenthesis indicate the time cycle of effluent flow through the tubes

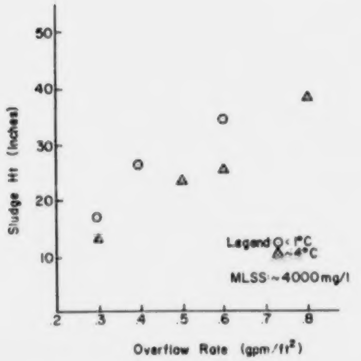


FIGURE 5 **8.9 gal. reactor sludge heights in effluent tubes vs. effluent overflow rates with continuous flow through tubes**

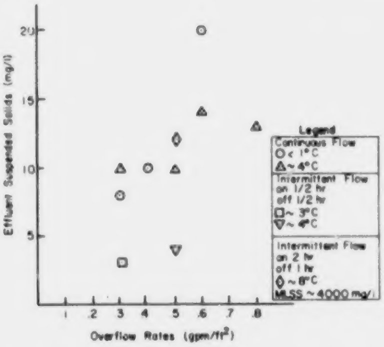


FIGURE 6 **8.9 and 12.45 gal. reactors effluent suspended solids vs. overflow rates at various temperatures**

an aerated lagoon and an extended aeration basin. The purpose of the facility was to increase the knowledge of biological waste treatment at cold temperatures and to develop design criteria.

Eielson Air Force Base is located 22 miles southeast of Fairbanks. The mean annual temperature at Fairbanks is approximately 25° F with minimum and maximum recorded temperatures of -66° F and +99° F respectively (Johnson and Hartman, 1969). The area has approximately 150 days below 0° F.

Originally intended to serve as a facultative lagoon, the extended aeration unit consisted of an earthen basin lined with 20 mil polyvinyl chloride film (PVC) and tube settler modules as shown in Figures 7 and 8. The PVC film at the bottom of the basin was covered with 6 inches of sand and a concrete pad poured in the center for support of aerators. Aeration and mixing were provided by eight Hydroshear aerators, manufactured by the Chicago Pump Company. Air supply was by a 120 SCFM Sutorbilt blower, manufactured by the Fuller Company.

Solids separation was provided by two tube settler modules. The tube settlers were developed by Neptune Microfloc Company for use in water treatment. The manufacturer has recently initiated studies to adapt them for use in activated sludge separation (Hansen, Culp and Stukenberg, 1967). This type of settler was felt to provide optimum design for submerged operation which was desired to overcome icing problems. The basin was fed by a Marlow centrigal pump, manufactured by ITT Marlow Company. Pumping rate was approximately 180 gpm with the feed drawn from a manhole on the influent line just before entry to the EAFB primary treatment plant. Temperature of the sewage averages about 20° C with the sewer lines enclosed in a utilidor, which is heated during the winter months.

The extended aeration facility was built to provide the very simplest operation with a minimum of environmental protection for evaluation under cold climate conditions. Construction of the extended aeration facility was completed in December 1968 and the unit placed in operation later that month. The unit was operated at a 2-day detention time, which corresponded to an average overflow loading rate on the tube settler of 1.3 gpm/ft^2. A problem was encountered with breakage of pumps due to entrained solids entering the pumping chamber. The feed line also filled with solid material and plugged. As a result, the basin was not fed for a week, during which time 3 feet of ice formed over the pond and frozen foam built up to 8 feet above the aerator.

Beginning in January 1969 and lasting approximately 6 weeks, a period of extremely cold weather occurred with ambient air temperatures dropping as low as -60° F. A detention time of 1 day was maintained during this period with no ice forming. The loading on the tube settlers was approximately 2.5 gpm/ft^2. The gear housing of a compressor was broken and teeth stripped from the gears while attempting to start it at a low temperature. Apparently, metal contraction had reduced clearances which caused internal rotating parts to make contact with and break the pump housing.

During February, the feed pumps were moved inside the Eielson primary treatment plant and feed taken from the grit chamber. For the remainder of the winter and the following spring, while operating at a detention time of 2 days, the MLVSS of the system generally did not rise above 500 mg/l.

Inadequate mixing was suspected as the cause of poor performance, and velocity measurements were made with an ice current meter obtained from the U. S. Geological Survey which measured the horizontal component only. Velocities were generally lower than the 1.0 ft per second recommended for complete mixing, except within 2 feet of the surface. The aeration rate was 120 cfm, depth of the basin 11 feet, with approximately 4 horsepower input. Velocities were measured again at a later date with 300 cfm being delivered and 9 horsepower input with generally the same results except the surface velocities were higher. The velocities found were not considered low enough to cause the extremely poor basin performance.

The possibility of excessive turbulence being carried into the tubes was also considered because of the close proximity of the settler modules to the aerators (2-3 feet). To check the possibility, a new aerator was fashioned of a short length of 3-inch pipe attached to flexible hose and placed in the basin approximately 10 feet from the settler modules. The MLSS of the basin increased to

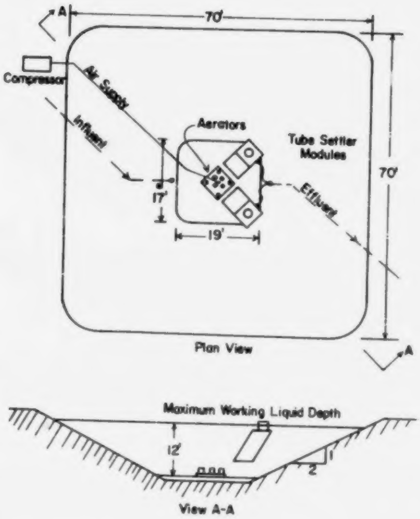

A
70'
Compressor
Air Supply
Influent
Aerators
Tube Settler Modules
Effluent
17'
19'
70'
Plan View
A
Maximum Working Liquid Depth
12'
1
2
View A-A

FIGURE 7 Extended aeration pilot facility, Eielson AFB (1968 configuration)

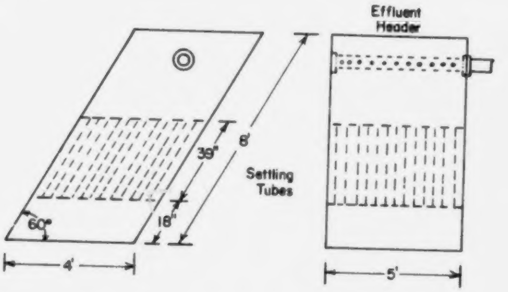
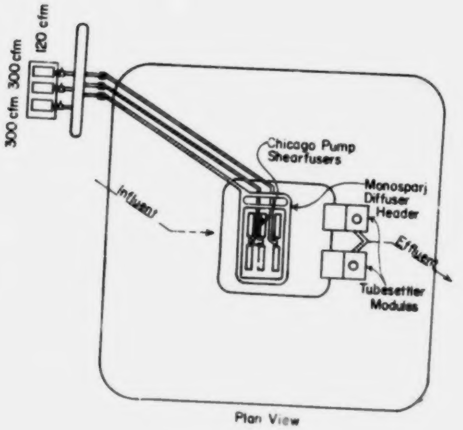

Effluent Header
8'
39"
Settling Tubes
18"
60°
4'
5'

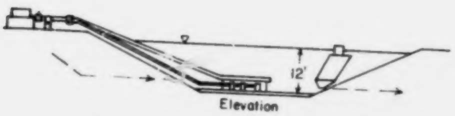
FIGURE 8 Tube settler module (1968 configuration)

1,000 mg/l during operation of this aerator which did indicate that basin turbulence or entrained air bubbles was affecting the settler operation.

The basin was then taken out of operation to permit modifications in preparation for the next winter's operation. The modifications are illustrated in Figures 9 and 10. The system was placed in operation in December 1969. It was recognized that at a detention time of 24 hours and with low winter operating temperatures, the hydraulic load on the tube settlers would be too great. An attempt was made to reduce the hydraulic load on the system while it maintained a BOD load equivalent to a 24-hour detention time system by supplementing the feed with primary sludge from the Eielson treatment plant. Basin velocity proved to be restricted around and beneath the


300 cfm
300 cfm
120 cfm
Chicago Pump Shearfusers
Monosparj Diffuser Header
Influent
Effluent
Tubesettler Modules

Plan View



Elevation

FIGURE 9 Extended aeration pilot facility after modification, Eielson AFB (1968 configuration)

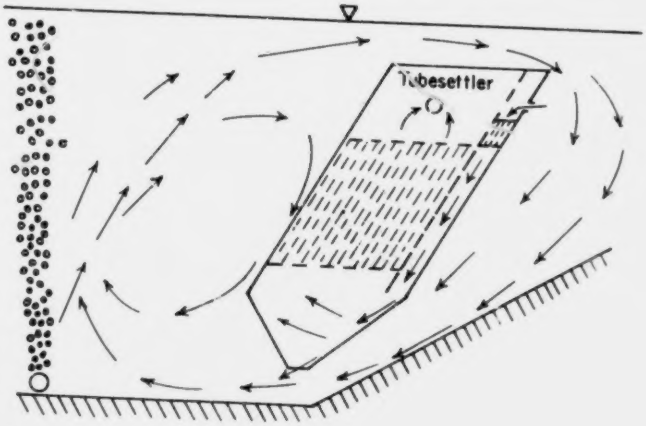
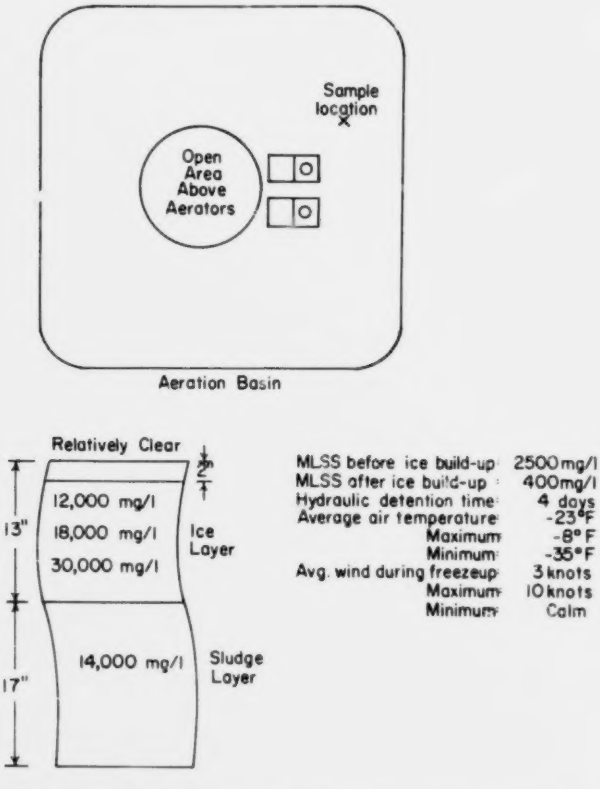

Tubesettler

FIGURE 10 Tube settler beneath module design with flow beneath the hopper (1969 configuration)

separator hoppers because of the low clearance and resistence offered by the settler support. As a result, a heavy sludge deposit blocked the separator hoppers, accumulated in the tubes and passed into the effluent.

During a cold period in January 1970, the surface of the basin began to freeze due to low heat energy being supplied. The sludge accumulated in the ice, reducing the suspended solids level in the pond from approximately 2,500 mg/l to less than 200 mg/l. During this period, the mean ambient temperatures averaged -23° F, with a range of -8° to -35° F. Wind velocity ranged from 10 knots to calm and averaged 3 knots.

A block was cut from the ice and a sample taken of the unfrozen sludge beneath the ice. A cross section is shown in Figure 11. The ice had reached a thickness of 14 inches with a sludge layer of 17 inches beneath the sampling point. The sludge was not moving under the ice and apparently had attached itself, building up a thicker and thicker layer which eventually froze into the ice layer.


Sample location
Open Area Above Aerators
Aeration Basin
Relatively Clear
2in
13"
12,000 mg/l
18,000 mg/l
30,000 mg/l
Ice Layer
17"
14,000 mg/l
Sludge Layer

MLSS before ice build-up:	2500mg/l
MLSS after ice build-up :	400mg/l
Hydraulic detention time:	4 days
Average air temperature:	-23°F
Maximum:	-8° F
Minimum:	-35°F
Avg. wind during freezeup:	3 knots
Maximum:	10 knots
Minimum:	Calm

FIGURE 11 Profile of surface ice and sludge layers of frozen extended aeration basin

The long sloping side walls associated with earthen basins present two very important problems in activated sludge aeration chamber applications. The relatively high surface area-to-volume ratio will result in high heat energy losses from the system which may be very critical with low temperature influent. Greater heat losses will promote ice formation which will entrain MLSS from the system, destroying the effectiveness of the process.

The second problem is the difficulty in obtaining adequate basin velocities at lower depths without excessively high horsepower for mixing. Even at high aeration rates, the minimum recommended velocities of 1.0 fps were generally not present in the pilot facility extended aeration basin at Eielson Air Force Base.

Another effect observed during operation in the second winter was that, with the aerators off center, a circular flow was induced in the basin in the horizontal plane around the aerators. The flow was similar to the Coriolis effect and seemed to be promoted by the earthen basin shape of a large surface area-to-bottom area ratio. This effect will only become a problem in situations in which flow directions in the basin are important, as in the Eielson AFB pilot facility, where the circular flow pattern did have an effect in hindering sludge removal from beneath the hoppers.

The cross sectional shape of a basin and the temperature to which it is exposed will, in general, determine the type of liner which should be provided. Material must be used which will prevent erosion and scouring by velocities in the basin. Side slopes of less than 1 vertical to 2 horizontal permit use of flexible liners, whereas vertical sides will require bearing wall construction of impermeable concrete or wood crib design with an impermeable liner.

Experience with the PVC liner indicates it is not feasible for use in permanent installations for cold temperature applications. The liner becomes very susceptible to damage at low temperatures because of brittleness, and ice formation can cause extensive breaks in the lining. Aging and exposure to sunlight also increase its susceptibility to damage.

Impermeable liners such as low temperature butyl rubber membranes are feasible for use in earthen basins when the danger of major freezing does not exist. Care must be taken to insure that the liner is resistant to hydrocarbons which may be present in the sewage as softening or dissolution may result.

Concrete provides a reliable material for cold temperature application. However, construction is expensive in Alaska and particularly so in remote areas. Examples of the successful application of cheaper methods of concrete construction are the College Utilities oxidation ditch in Fairbanks, Alaska, and the oxidation ditch at Glenwood, Minnesota (Anonymous, 1965). Concrete block was used for the construction of vertical sides for the College Utilities ditch. Concrete silo staves were originally used for the sloping sidewall construction of the Glenwood ditch but were not sealed and soil behind the staves washed out. The problem was successfully alleviated by placing steel mesh and 4 inches of concrete grout over the staves to provide a smoother waterproof lining.

CONCLUSIONS

The feasibility of the extended aeration activated sludge process as a relatively economical and effective means of secondary waste treatment has been demonstrated in the laboratory and in the field. The process requires more consistent operation and maintenance than aerated lagoons and this is a disadvantage where costs are high and skilled operators are extremely scarce.

The utilization of exposed aeration chambers for the extended aeration process is feasible. Earthen basins are also feasible for use where economic and construction conditions warrant. When utilizing exposed basins, heat loss effects must be evaluated in conjunction with detention time determinations to avoid potential freezing problems. Solids entrainment in ice can cause failure of an activated sludge process.

Environmental protection in varying degrees should be provided for the remaining equipment, such as heated enclosures for pumps and flow measurement devices. Housing must be provided for secondary sedimentation basins and should include a minimum of an unheated structure with panels which can be removed for warm weather operation.

Effective solids separation is the key to successful operation of extended aeration facilities and is dependent on both the biological and physical aspects of the system. It has been demonstrated that a sludge can be developed which will perform very efficiently at temperatures less than 1° C.

A turbid effluent will result at cold temperatures with an unacclimated sludge or loading rates that are too high. Under these conditions, a less stabilized sludge develops with a corresponding relative decrease in numbers of stalked ciliates and an increase of dispersed bacteria which appears to contribute to turbidity (McKinney and Gram, 1956).

A bulking sludge may develop at cold operating temperatures. This type of sludge can lead to separation problems but will provide a very clear effluent at temperatures ranging down to less than 1° C.

Properly designed tube settlers will provide effective cold (0 to 4° C) temperature solid separation. This is true for sludges with SVI's ranging up to 250. A backwash cycle should be provided for reliable operation and is mandatory for operation with high MLSS concentration (4,000 mg/l) and bulking sludges. Some effort should be directed toward developing upflow clarifier configurations for cold temperature application since the method has advantages (Reed and Murphy, 1969). The tube settler does provide an upflow clarifier type action in high MLSS activated sludge solid separation applications. Providing consistent solids separation with tube settlers at warmer temperatures (greater than 4° C) appears to be the most demanding and yet insufficiently defined area of need in their application.

Cold climate sludge wasting and disposal for the extended aeration process must be given consideration for the following reasons:

1. Excess solids production increases with decreasing temperature.

2. Shorter detention times to prevent freezing will also increase solids production at a given MLSS level.

3. Auto-induced sludge wasting may be expected to be more severe, placing greater potential stress on the receiving water.

4. Retarded assimilative capabilities of the receiving water at cold temperatures.

RECOMMENDATIONS

Facility Design

The following design recommendations are based on laboratory studies, experience with the Eielson Air Force Base pilot facility, and experience reported by others.

(1) Exposed aeration basins should be considered for reducing construction costs of waste treatment facilities. Raw sewage temperatures and heat loss effects must be considered to prevent freezing which can cause process failure by entrainment of solids from the system.

(2) Housing should be provided for pretreatment units such as bar racks, pumps and flow measuring equipment.

(3) Some minimum protection should be provided for aeration equipment such as strip heaters or minimum heat enclosures for compressors and unheated housing for oxidation ditch rotors.

(4) Housing should be provided for secondary sedimentation basins. Minimum housing would include a structure with panels which may be removed for warm weather operation.

(5) Where economic and construction considerations warrant, earthen basin designs should be considered for aeration chamber construction. Otherwise, sidewalls that are vertical or nearly so should be utilized to promote better mixing.

(6) Submerged settling units should be situated in the center of basins with low sidewall construction with aeration on at least two sides to promote adequate mixing. Several questions require answers before submerged settling units are practical.

(7) When basins with low sidewall slope construction are utilized without submerged settling units, the aeration devices should be clustered in the center of the basin for best mixing.

(8) Flexible membranes should not be used where the danger of heavy icing exists.

(9) Concrete block and concrete grout should be considered as economical liner materials where the design permits.

(10) Tube settlers with backwashing of tubes should be considered. However, more information is necessary before their reliability can be ascertained. Both routine and emergency maintenance must be carefully evaluated prior to their use.

Process Design

The following preliminary recommendations for low temperature extended aeration systems are based on laboratory studies and experience reported by other investigators. Attempts will be made to verify these findings on a pilot plant scale.

(1) Organic loadings should be maintained below 0.20 lb. BOD/lb. MLVSS-Day.

(2) Provision should be made for sludge wasting of 0.5 lb. MLSS/lb. BOD removed, particularly at shorter detention times such as a 12-hour detention time system.

(3) Tube settler overflow rates should be held below 0.5 gpm/ft^2 with high MLSS concentrations ($>$ 4,000 mg/l).

(4) Sludge wasting and disposal facilities or a polishing lagoon for effluent discharge should be provided where heavy discharges of suspended solids may place excessive stress on receiving waters.

Research and Development Needs

The following list of suggested research and development needs is not intended to be all inclusive but includes areas which have come to the attention of the authors through laboratory and pilot plant experience and a review of experience reported by other investigators:

(1) Sludge bulking conditions at lower temperatures (8° C and below) must be defined so the condition can be predicted in actual application.

(2) Further develop low temperature biokinetic parameters at detention times ranging from 4 to 36 hours with varying MLSS levels.

(3) Further develop low temperature tube settler design criteria and backwashing techniques at various MLSS levels.

(4) Investigate upflow clarifier designs for low temperature application.

(5) Investigate methods of sludge digestion and disposal under low temperature conditions; particularly the use of the freeze-thaw cycle as an aid to promoting better drainability.

(6) Develop reliable methods for positive recirculation of settled solids from submerged settling units.

(7) Further investigate criteria for predicting heat loss from exposed basins.

(8) Continue evaluation of cold temperature biokinetic design parameters on pilot plants and existing facilities.

(9) Develop design and operation criteria for low temperature horizontal flow clarifiers.

(10) Investigate power requirements and mixing characteristics of various earthen basin configurations.

(11) Further investigate the effects of heavy ice cover on solids entrainment in aeration basins, particularly where flow patterns are parallel to the surface as in the oxidation ditch.

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CHEMICAL TREATMENT OF MECHANICALLY AND BIOLOGICALLY TREATED WASTEWATER

Arne Rosendahl

INTRODUCTION

Eutrophication problems are found to be serious in many Norwegian lakes and fjords.

An extensive survey of the pollution in the inner Oslofjord has shown that organic matter produced by autotrophic growth, mostly due to the discharge of nutrients, is 10 - 12 times the primary organic load from wastewater. In this stratified, landlocked fjord, with a brackish surface water, the nutrients found to be most important are phosphorus, nitrogen, and iron. With the large amounts of nitrogen and iron naturally being brought into the system, phosphorus is at the present the only controllable component.

The same reasoning is found applicable to most of our eutrophied waters, and numerous treatment plants for removal of phosphorus from wastewaters will have to be installed in years to come.

Today phosphorus removal is thought to be most economical by chemical coagulation/flocculation and with sedimentation or flotation for the liquid/solid separation.

Several people have studied the theoretical basis for such chemical processes (Stumm and Morgan, 1962; Stumm, 1962; Pöpel, 1966; Henriksen, 1962; and Henriksen, 1963) but there is still a lack of some essential knowledge. This makes it difficult to make a process design on a purely rational basis, and the treatment plants now in operation are therefore designed on experience gained through chemical treatment of drinking water during many years, and through experiments made in new plants for phosphorus removal (Wuhrmann, 1964).

The great majority of these plants are built as tertiary units, i.e. chemical treatment of biologically treated waste. At a few plants chemical and biological treatment take place simultaneously in the same reactor (Thomas 1962).

In Norway there are only a few biological treatment plants, and we have therefore been looking for a more direct way of phosphorus removal by omitting the biological unit. During 1967 some treatment plants were built in Sweden using mechanical/chemical treatment, but systematic observational data describing the process was limited.

Pilot plant studies were needed giving comparable results between mechanical/chemical and biological/chemical treatment systems, and results and experience that could be translated to full-scale treatment plants. In 1968 a study was started at the Norwegian Institute for Water Research, and results from 1½ years' program are reported in this paper.

DESCRIPTION OF THE PILOT PLANTS

Five pilot plants were used for this investigation. They were situated at the site of a biological treatment plant owned by the city of Oslo, treating the effluent of about 50,000 inhabitants, using the conventional activated sludge method.

One extended aeration plant, type Hycon 5 A, previously used by this institute to study kinetics of biochemical treatment, was reconstructed. This plant was divided on four parallels based on coagulation/flocculation and sludge removal by sedimentation (Fig. 1). A fifth separate plant used coagulation/flocculation and sludge removal by flotation (Fig. 2).

All five plants were in use from January 1969 until May 4, 1970. On that day a serious fire stopped the experiments at the four sedimentation plants.

Each plant consisted of:

1. Wastewater intake arrangement
2. Chemical feeding equipment
3. Rapid mixing
4. Flocculation basins
5. Sludge-separation equipment (sedimentation or flotation)
6. pH recorder
7. Sampling equipment

Sedimentation Plants

During this investigation primary treated wastewater was used in two plants and secondary treated wastewater was used in the two other plants.

During the first investigation period a technical quality of aluminum sulfate, type AVR from Boliden i Sweden, was used. (This type of alum has about 2% impurities, mainly iron and silicates. It is produced for chemical treatment of sewage and is at the moment the cheapest alum in Scandinavia). In the last part of the investigation period sulfuric acid was dosed in addition to alum, for pH adjustments. Both alum and sulfuric acid were made up to a 5% solution. The chemicals were added to the wastewater ahead of the flocculation basin. To obtain sufficient mixing the chemicals were added where the water was quite turbulent.

The flocculation basin had four treatment plant paddles mounted on horizontal parallel axes. They were driven by the same motor, and ran at the same speed. The speed, however, could be varied within a wide range. Each flocculation basin was 2.5 m long, 0.6 m wide and 0.8 m deep, and was divided into three chambers by vertical walls with a 30 cm hole around the paddle axes to prevent short circuiting. Periphery velocity was selected as 0.4 m/sec. This gives theoretically calculated

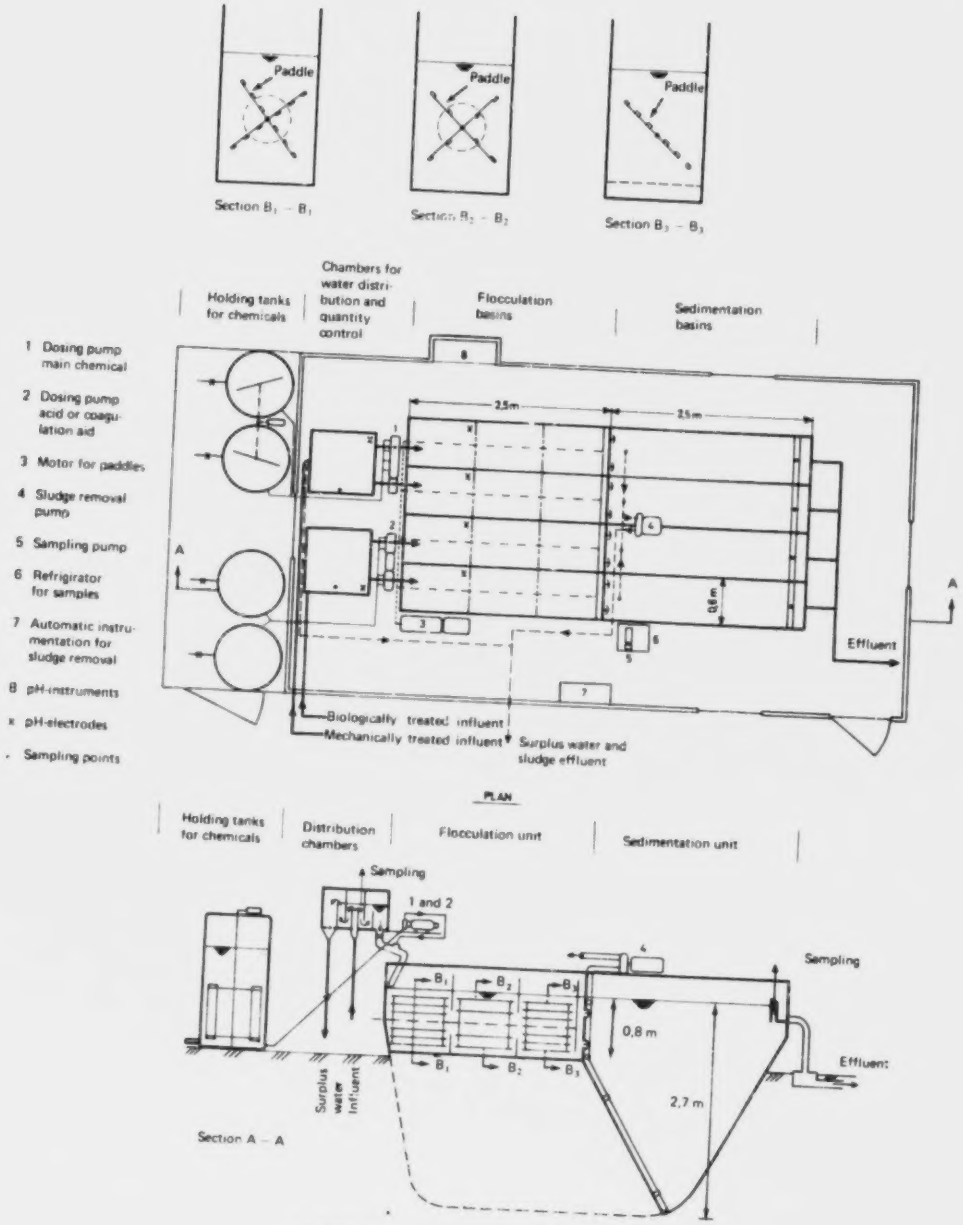
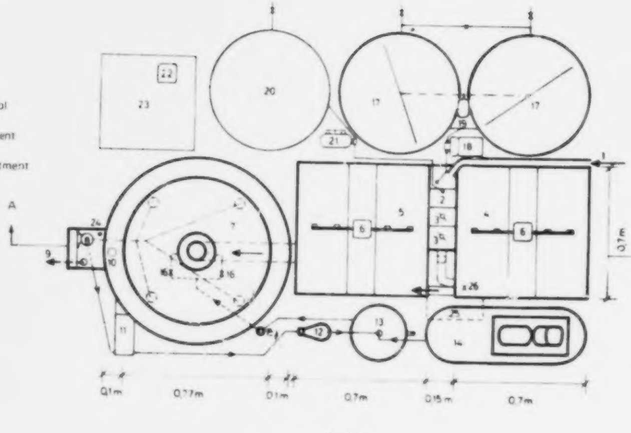

Paddle
Section B1 – B1
Section B2 – B2
Section B3 – B3
Holding tanks for chemicals
Chambers for water distribution and quantity control
Flocculation basins
Sedimentation basins
1 Dosing pump main chemical
2 Dosing pump acid or coagulation aid
3 Motor for paddles
4 Sludge removal pump
5 Sampling pump
6 Refrigirator for samples
7 Automatic instrumentation for sludge removal
8 pH-instruments
x pH-electrodes
· Sampling points
2,5m
2,5m
0,8m
A
A
Effluent
Biologically treated influent
Mechanically treated influent
Surplus water and sludge effluent
PLAN
Holding tanks for chemicals
Distribution chambers
Flocculation unit
Sedimentation unit
Sampling
1 and 2
4
Sampling
B1
B2
B3
0,8 m
2,7 m
Effluent
Surplus water
Influent
Section A – A

FIGURE 1 Sedimentation plants

1	Influent
2	4 rapid mix chamber
3	Rapid mixer for scum control
4	First flocculation compartment
5	Second flocculation compartment
6	Paddle with motor
7	Flotation unit
8	Shut off valve for effluent
9	Effluent
10	Sludge outlet
11	High pressure pump
12	Level regulator
13	Pressure tank
14	Compressor


A
A
1
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3
4
5
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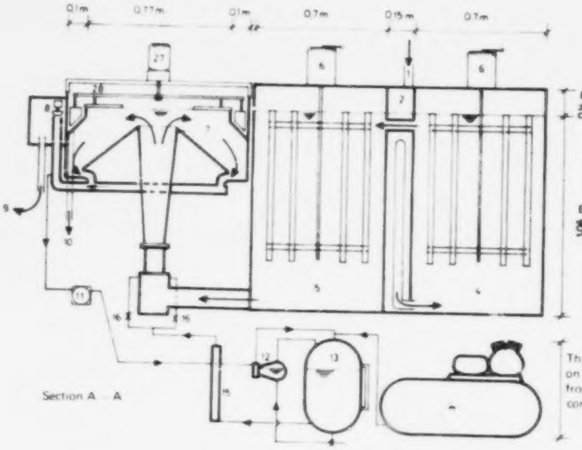
16
17
18
20
21
22
23
24
25
26
0.7m
0.1m
0.77m
0.1m
0.7m
0.15m
0.7m

PLAN

15	Rotameter
16	Reduction valve for dispersion
17	Holding tanks for main chemical
18	Dosing pump for 17
19	Motor and paddle for chemical holding tanks
20	Holding tank for acid or coagulation aid
21	Dosing pump for 20
22	Sampling pump
23	Refrigirator for samples
24	Sampling point
25	pH instrument
26	pH electrode
27	Motor for sludge scraper
28	Sludgescraper


0.1m
0.77m
0.1m
0.7m
0.15m
0.7m
0.16m
1.06m
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16
16
27
28

Section A – A

This equipment is placed on the same level in front of the flocculation compartments

FIGURE 2 Flotation plant

g-values of 59 sec^{-1}, 55 sec^{-1} and 46 sec^{-1} in the first, second and third chambers (Camp, 1943). The influence of g-value variation at the plant has been studied as a diploma thesis for Civil Engineering in the autumn 1969.

The sedimentation units each had a length of 2.5 m, the width being 0.6 m. The bottom had steep slopes to the middle and the deepest point was 2.7 m below the surface. Water entered the basin from one end, flowing about horizontally through the basin and leaving at the other end over a V-notch weir. Sludge was withdrawn automatically for 30 seconds every hour.

Equipment for pH recording was installed in autumn 1969. and pH values were recorded on both mechanically and biologically treated wastewater before adding chemicals, and in the first flocculation chamber of each plant. Sampling was done continuously during every 24 hour period, and the bottles placed in a refrigerator.

The hydraulic load was 1.2 m^3/hr (5.28 gal/min) to each unit. This load was selected after trial runs with varying loads. Sludge loss from the sedimentation units was kept at a practical level. This load gave a theoretical detention time of 1 hr in the flocculation units, about 1.5 hr detention time, and theoretical overflow rate of 0.8 m/hr (470 GPD/Sq Ft) in the sedimentation units.

The Flotation Plant

Primary effluent was treated in the fifth plant throughout the whole investigation period. 10% solution alum as well as sulfuric acid was used.

The chemicals used were mixed with the wastewater in a baffled rapid mixing basin. The plant was provided with two flocculation chambers coupled in series, with separate paddles on vertical axes. Each flocculation chamber had an area of 0.7 x 0.7 m and a depth of 1.06 m. Paddles ran with a peripherical speed of 58.5 cm/sec in the first chamber and 35 cm/sec in the second chamber. This gave theoretical g-values of 78 sec^{-1} and 37 sec^{-1} in the first and the second chamber.

The flotation unit itself was circular with a diameter of 97 cm and a depth of 37.5 cm, made of concentric cylinders and cones. Flotation was effected by small air bubbles. They were produced by solute air in water under 5 atm. pressure. The pressure was released just before adding the flotation medium to the flocculated wastewater.

The sludge floated at the surface while the water flowed downwards and left the unit through four pipes at the periphery. The four pipes were connected to a larger pipe. The water level in the unit was controlled at the outlet. About 15% of the effluent was used to make up the dispersion water. This quantity was manually controlled by a flowmeter.

Sludge withdrawal was automatically controlled by a timer. Sludge was removed by closing the outlet valve for approximately 2 minutes every hour. While the water level increased, sludge and water left the plant over a mechanically cleaned weir.

The flotation plant was designed for 2 m^3/hr (8.8 gal/min) loading. This hydraulic load was kept constant throughout the test period. This gave a total theoretical detention time of 31 minutes in the flocculation chambers. Theoretical detention time in the flotation chamber was 5.1 minutes with the overflow rate of 4.3 m/hr (2520 GPD/Sq Ft). The plant had a pH recorder. Composite sampling was done during the complete test. Samples of the raw wastewater entering the community plant were also analyzed.

ANALYZED PARAMETERS

Several parameters were considered for the evaluation of the quality of the influent, effluent and kinetics of the processes.

The following parameters were analyzed:

Organic matter: chemical oxygen demand (COD) by the dichromate method

Biochemical oxygen demand, seven days test (BOD_7)

Total phosphorus, orthophosphate

Nitrogen: Kjeldahl and the sum of nitrite-nitrate nitrogen.

Turbidity (JTU)

pH, continuously recorded in flocculation chamber, and measured on day-samples in the laboratory.

Aluminum

TABLE 1

Mechanical/Chemical Treatment

Sludge Separation by Sedimentation

Chemical Dosed: Alum

			Parameters					
Water * quality	Chemical dose alum mg/l	No. of day samples for ave. value	COD ($K_2Cr_2O_7$) mg O/l	BOD_7 mg O/l	Tot. P mg P/l	Kjeldahl N mg N/l	Turb. JTU	pH
R	-	4	186.4	94.5	4.65	19.2	61.0	7.30
M	-	4	166.6	48.0	4.58	20.9	28.5	7.11
M/C	75	4	91.1	35.0	3.75	17.5	15.8	7.12
R	-	8	158.6	98.0	4.45	17.3	51.8	7.56
M	-	6	104.3	49.6	4.13	19.2	14.6	7.39
M/C	100	5	59.0	18.3	1.61	13.2	10.8	6.86
R	-	4	186.4	94.5	4.65	19.2	61.0	7.30
M	-	4	166.6	48.0	4.57	20.9	28.5	7.11
M/C	125	4	42.1	10.5	0.14	17.6	1.5	6.74
R	-	8	158.6	98.0	4.45	17.3	51.9	7.56
M	-	6	104.0	49.6	4.13	19.2	14.6	7.39
M/C	150	5	44.0	17.5	0.14	14.3	0.9	6.42

Calcium, magnesium and alkalinity were measured on a series of samples to obtain the average hardness of the treated water.

The process variables selected were chemical dosage, hydraulic loading and flocculator speed (g-values).

RESEARCH PROGRAM AND RESULTS

The first research series covered doses of 75, 100, 125 and 150 mg/l alum. These doses were added to biologically treated wastewater and mechanically treated wastewater.

The results are given in Tables 1, 2 and 3, and some results are shown graphically in Figure 3.

During these experiments it was obvious that quality of the effluent varied extremely during the day. The results seemed to follow the variations of pH which could vary within ± 1.0 pH unit during the day, with the lowest pH at 6:00 a.m. and the highest value about 1:00 p.m. Turbid and colored effluent was noticed when the pH was higher than 6.2 - 6.5 in the flocculation chamber. It looked as if a critical pH value was lower at a lower dose of alum. From this it was decided to try alum doses of 70, 80, 90 and 100 mg/l and adjust pH to a maximum of 6.5 using sulfuric acid. The results of these experiments are listed in Table 4 and some results are shown graphically in Figure 4.

TABLE 2

Mechanical/Chemical Treatment

Sludge Removal by Flotation

Chemical Dosed: Alum

			Parameters					
Water * quality	Chemical dose alum mg/l	No. of day samples for ave. value	COD ($K_2Cr_2O_7$) mg O/l	BOD_7 mg O/l	Tot. P mg P/l	Kjeldahl N mg N/l	Turb. JTU	pH
R	-	10	231.4	122.0	5.04	21.5	96.1	7.23
M	-	8	149.0	58.4	4.63	20.9	26.6	7.05
M/C	75	10	92.4	25.8	2.84	19.2	20.0	7.07
R	-	4	137.5	83.5	4.43	16.9	29.9	7.54
M	-	4	101.4	47.0	3.98	19.7	13.3	7.44
M/C	100	4	45.8	14.8	0.99	15.2	5.6	7.04
R	-	8	209.1	114.0	5.88	28.1	34.0	7.50
M	-	9	123.9	47.0	4.53	20.2	16.2	7.27
M/C	125	9	47.4	10.9	0.32	19.9	1.1	7.02
R	-	7	194.3	132.8	4.68	19.5	62.4	7.46
M	-	5	109.5	40.3	4.38	17.5	15.9	7.14
M/C	150	7	34.4	8.7	0.19	15.6	1.3	6.61

*
R = Raw wastewater
M = Mechanically treated
B = Biologically treated
C = Chemically treated

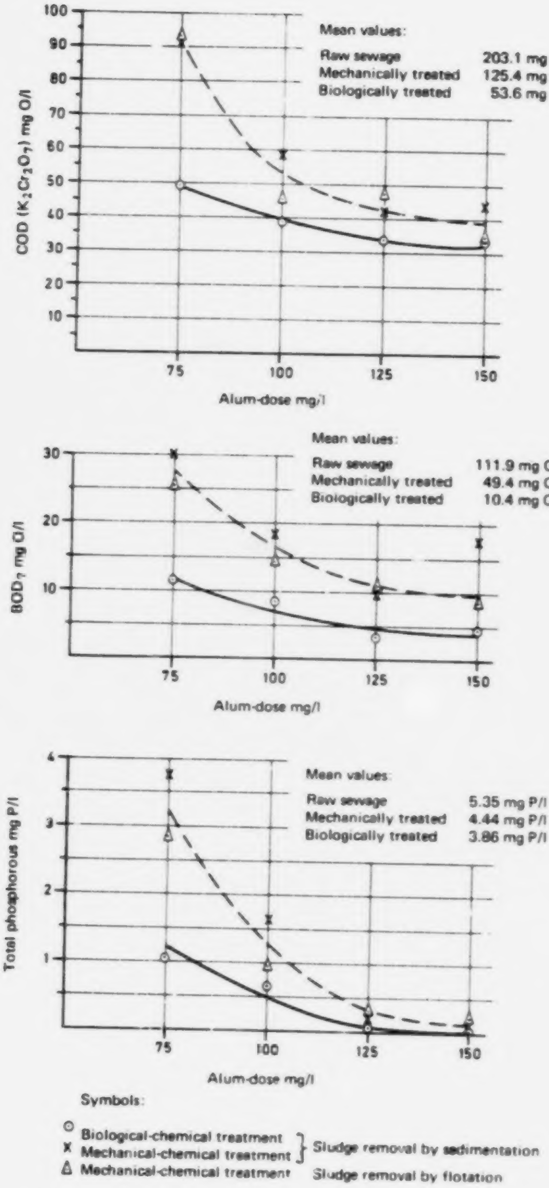

FIGURE 3 Results by dose of alum at sedimentation and flotation units on mechanically and biologically treated wastewater

The pH values given in the tables are average values of daily samples measured in the laboratory. They are approximately 0.5 pH units higher than the value measured in the flocculation chamber. The influence of pH will be studied further on the fifth unit (using flotation for sludge removal) where a full pH control will be carried out.

The values in Tables 1 to 4 are average for several 24 hr-samples. The samples are mostly from a period of about 14 days. The number of samples from which the average values were calculated varied from 3 to 10. The series with the lowest numbers were supposed to be redone. The points plotted in the diagrams are from the tables.

Phosphorus Removal

From January until May 1969 the phosphorus content of the raw waste averaged 5.35 mg/l P. Mechanically treated water from the community plant had in the same period an average of 4.44 mg/l P and the biologically treated water had 3.86 mg/l P.

The curves in Figure 3 show an increase in phosphorus removal with an increasing dose of 75 to 125 mg/l alum for both mechanically as well as biologically treated wastewater. From 125 to 150 mg/l there is only a slight increase in the quality of the effluent regarding phosphorus removal.

At the dose of 75 and 100 mg/l with alum there is a significant difference in quality of the effluent between biologically and mechanically pretreated waste used. Thus, effluent with about 1 mg/l P can be achieved when treating biologically pretreated wastewater and 3 - 3.5 mg/l P when treating

TABLE 3

Biological/Chemical Treatment

Sludge Removal by Sedimentation

Chemical Dosed: Alum

			Parameters					
Water * quality	Chemical dose alum mg/l	No. of day samples for ave. value	COD ($K_2Cr_2O_7$) mg O/l	BOD_7 mg O/l	Tot. P mg P/l	Kjeldahl N mg N/l	Turb. JTU	pH
R	-	4	186.4	94.5	4.65	19.2	61.0	7.30
B	-	4	59.1	12.0	4.00	17.1	4.3	7.31
B/C	75	4	49.3	11.5	1.08	15.9	2.7	6.98
R	-	7	154.2	97.0	4.45	17.3	47.5	7.59
B	-	5	49.3	11.6	3.68	14.5	1.7	7.31
B/C	100	4	38.8	8.5	0.70	14.0	1.4	6.76
R	-	4	186.4	94.5	4.65	19.2	61.0	7.30
B	-	4	59.1	12.0	4.00	17.1	4.3	7.31
B/C	125	4	33.3	3.5	0.07	15.1	0.5	6.93
R	-	8	158.6	98.0	4.45	17.3	51.8	7.56
B	-	6	50.6	10.0	3.77	14.9	2.0	7.29
B/C	150	6	34.1	4.7	0.09	14.1	0.3	6.58

mechanically pretreated wastewater using a chemical dose of 75 mg/l alum. At a dose of 100 mg/l alum about 0.5 mg/l 0 and 1 - 1.5 mg/l P can be obtained using chemical treatment respectively.

At the dose of 125 mg/l alum the results vary from 0.07 to 0.32 mg/l P, and at the dose of 150 mg/l from 0.09 to 0.19 mg/l P. Again the best results using chemical treatment were achieved with biologically pretreated wastewater. At these doses the difference between the quality of the wastewater, pretreated mechanically or biologically, is not significant.

Using acid to lower the pH caused a marked difference in the quality of the effluent from 70 to 90 mg/l alum. From 90 to 100 mg/l alum there was only a slight increase in the quality.

It seems as if the same results can be obtained using a dose of 90 - 100 mg/l alum and pH control compared to a dose of 125 mg/l alum without pH control regarding phosphorus removal.

Orthophosphate removal followed the trend of total phosphorus removal very closely.

Removal of Organic Matter

BOD_7 and COD of the influent and effluent from the chemical treatment plants, as well as the raw water entering the community plant, were measured. The curves and tables show that BOD and COD removals follow each other very closely.

TABLE 4

Mechanical/Chemical Treatment

Sludge Removal by Flotation

Chemicals Dosed: Alum and Acid

	Chemical dose			Parameters					
Water * quality	Alum mg/l	H_2SO_4 mg/l	No. of day samples for ave. value	COD ($K_2Cr_2O_7$) mg O/l	BOD_7 mg O/l	Tot. P mg P/l	Kjeldahl N mg N/l	Turb. JTU	pH
R	-	-	8	172.1	98.4	4.37	18.5	30.1	7.66
M	-	-	8	199.0	132.3	5.61	22.9	45.9	8.03
M/C	70	78	8	61.0	17.6	2.41	17.9	5.3	5.11
R	-	-	5	201.1	87.0	5.04	22.6	48.4	7.57
M	-	-	5	193.3	104.5	5.76	28.0	109.0	7.58
M/C	80	54	5	60.0	18.0	1.74	21.1	6.7	6.48
R	-	-	4	155.0	91.8	4.75	20.8	54.0	7.29
M	-	-	4	156.2	125.5	4.68	19.8	40.1	7.28
M/C	90	42	4	39.4	15.5	0.56	16.7	2.5	6.72
R	-	-	3	178.6	102.0	4.90	21.8	31.7	7.24
M	-	-	3	132.5	90.5	5.00	22.5	22.2	7.32
M/C	100	30	3	31.6	7.3	0.25	16.2	0.7	6.72

R = Raw wastewater
M = Mechanically treated
B = Biologically treated
C = Chemically treated

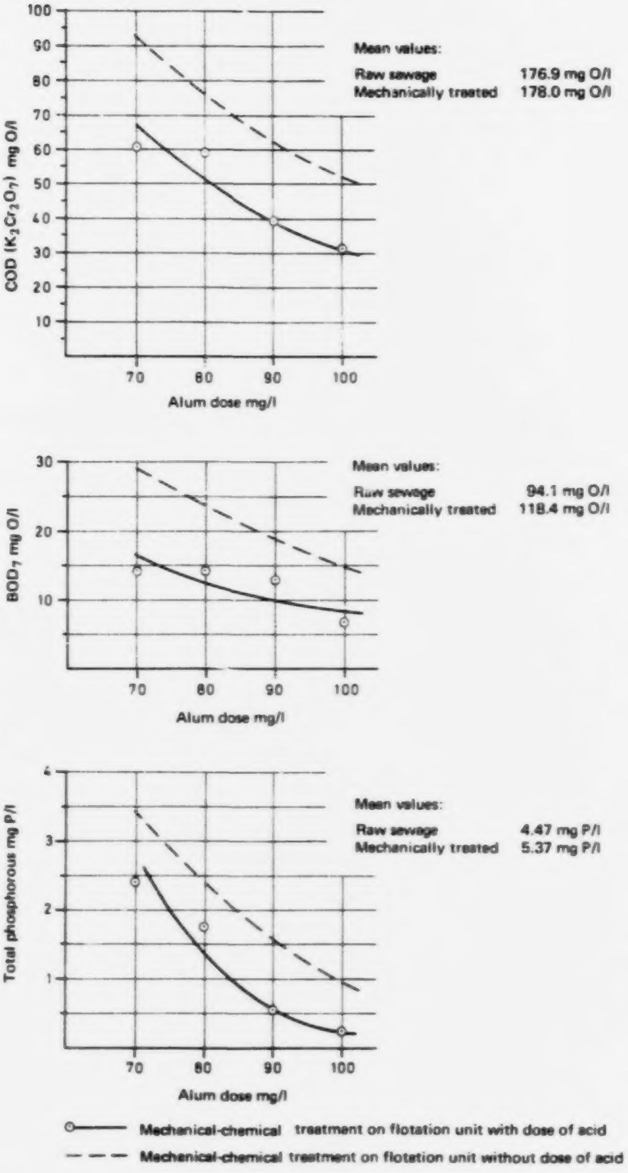

FIGURE 4 Results by dose of alum and acid at flotation unit on mechanically treated wastewater

The difference in the removal is larger between mechanical/chemical treatment and biological/chemical treatment for organic matter compared to phosphorus removal. This is also true when a high alum dose is used.

A dose of 75 mg/l alum gives only a slightly lower BOD and COD with chemical treatment. An increase in dose up to 125 mg/l alum gives better removal, especially on mechanically pretreated waste. An increase in chemical dose from 125 to 150 mg/l alum gives only a small improvement in organic matter.

It is most important that mechanical/chemical treatment using a dose of 125 to 150 mg/l alum gives an effluent regarding organic matter which has the same quality as biologically treated wastewater. From the curves and tables it is shown that biological treatment gives a COD reduction from 203 to 53 mg/l, and BOD_7 reduction from 112 to 10 mg/l. With a dose of 125 mg/l alum using mechanically treated water, an effluent of 45 mg COD and 11 mg/l BOD was achieved.

The results from a combined dose of 100 mg/l alum and pH adjustments are somewhat better than using 125 mg/l alum without pH adjustments. The curve is steeper in the range of 90 - 100 mg/l alum with acid than it is in the range of 125 - 150 mg/l alum without acid. It seems therefore that a higher dose of alum could give better results regarding organic matter when pH adjustments are made.

Other Components

Removal of turbidity followed phosphorus removal very closely. Aluminum residual was measured in the effluent and was thought to be a good parameter to control the process, but experience showed that turbidity and pH gave better information on the conditions of the plant, and were easier to observe. Turbidity and pH seemed to be the parameters most likely to indicate the efficiency of the process.

Nitrogen has been analyzed as Kjeldahl-nitrogen and as a sum of nitrite-nitrate nitrogens. Reduction of Kjeldahl-nitrogen was moderate by chemical treatment when using alum as a coagulant both with or without control of pH. There were only slightly better results when the chemical dose was increased. The amount of nitrogen was reduced from 20 - 15 mg/l N in the most effective experiments.

Nitrite-nitrate has always been below 1 mg/l N in the raw mechanically and biologically treated water. By the chemical treatment there has always been a tendency for higher values of nitrite-nitrate in the effluent compared to the influent for mechanically and biologically pretreated wastewater. However, the values have always been lower than 1 mg/l N except in two cases. Nitrite-nitrate always makes up less than 5% of the total nitrogen.

Average value of hardness for the wastewater treated was 1.7 dH. The experiments using alum and acid show higher values of most parameters for mechanically treated water as for the raw water. The reason for this has not yet been found. It may have a connection with the accumulation of suspended solids in the return surplus sludge from the biological units to the mechanical units.

The type of alum used so far is delivered at a price of 350,- N.kr/ton in Oslo. The price for sulfuric acid is 225,- N.kr/ton in 95% concentration. The prices are:

125 mg/l alum 4.37 øre/m^3
100 mg/l alum + 30 mg/l H_2SO_4 4.21 øre/m^3
(1 N.kr = 100 N.øre = $0.14)

The difference in price when using one or two chemicals is small. The prices will differ for different places in Norway, and other countries may have different prices. Treatment efficiency and economical aspects should be evaluated in each case.

Sludge production was measured, but is not presented in this paper. It is obvious, however, that sludge production decreases with decreasing doses of alum. This may be a deciding factor for choosing two chemicals rather than one.

It will probably be unusual to use two chemicals at smaller treatment plants. (In Norway there will be a large number of treatment plants serving less than 5,000 persons). The lower cost and smaller sludge production by using two chemicals will not outweigh the extra operational difficulties.

At larger treatment plants, however, reduction in operational costs is more important. It is more economical to invest more in equipment and automatic control of chemical dosing to reduce the cost of operation. Reduction in sludge production will also be more important at large plants. At such plants one will get full advantage through the combined dose of alum and acid, if prices are such that this combination gives the lowest running cost.

Flotation is not suggested for small scale treatment for economical and operational reasons. This process has more mechanical equipment and requires highly skilled personnel. Effluent to be used as dispersion water should have good quality to avoid clogging.

For larger plants these reservations should be easily overcome. Using flotation decreases the construction cost even though the process needs more mechanical equipment.

The type of alum used for the experiments contains some very fine insoluble impurities. These have a tendency to settle in pipes before they are mixed with the wastewater, thus clogging valves and pumps. They may also cause wear on pumps. The company Boliden, which is delivering this alum, has made suitable equipment for dosing the chemical.

It is assumed that chemical treatment is less affected by low temperatures than biological treatment. Therefore the effect of temperature was not considered during these experiments. The testing was run under the climatic conditions in Oslo which are not extreme. Outdoor temperatures may be 20 - 25° C below zero for about one month. The wastewater temperature is decreased to about 5° C.

Construction costs for biological and chemical treatment is evaluated by Dr. Wuhrmann (1967) to be about 30% higher than conventional biological treatment by the activated sludge method. The costs for mechanical/chemical treatment plants were estimated to be about 20% less than the costs for conventional biological treatment plants. Such a consideration will be valid for treatment plants larger than about 5,000 persons. Costs of smaller chemical treatment plants can be higher than biological treatment plants.

With the results from the experiments regarding removal of phosphorus and organic matter, it seems that mechanical/chemical treatment should be considered first where eutrophication is the greatest problem in the receiving waters and phosphorus is the most important nutrient to be removed from the wastewater.

FUTURE INVESTIGATIONS

We had an extensive program for future investigations using the five plants. It was interrupted by the fire in May, when the four sedimentation plants were totally damaged. There is a great need for further investigations, and the work will be continued.

The experiments using the flotation plant will be continued with Norwegian alum. pH equipment will be enlarged to cover automatic dosing of chemicals through pH control. Studies will then be continued to find the most suitable and economic operational conditions using alum and acid.

Further, it is of great interest to find practical methods for using ferro-sulfate as a coagulant. This is an industrial waste product and will probably be the cheapest coagulant in Norway. From laboratory studies (Henriksen, 1962 and 1963) it seems that it should be an easy and inexpensive way to oxidize the iron from di-valent to tri-valent before or during the coagulation process.

We are also interested in studying coagulation using lime and stripping of ammonia by aeration at high pH.

All investigations will be preferably carried out simultaneously on mechanically and biologically treated wastewater. We are also interested in studying direct chemical treatment on raw wastewater.

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BIOLOGICAL AND CHEMICAL WASTE TREATMENT EXPERIMENTS IN FAR NORTHERN SWEDEN

Peter Balmér

INTRODUCTION

The town of Kiruna is situated at 68° north latitude (Fig. 1) and is hence the northernmost town in Sweden. In the winter, temperature extremes of -35° C (-30° F) occur.

The existing primary treatment plant was overloaded and the town planned to replace it with a new plant with primary and activated sludge treatment.

The know-how of design of an activated sludge plant for treating low temperature-wastes was very limited at this time (1962 - 1963). In a literature review of Hilmer, it was concluded that laboratory data clearly showed that the metabolizing activity of the activated sludge was seriously hampered at low temperatures. Data on BOD reduction in activated sludge systems at low temperatures were partly contradicting, and obviously did not provide sufficient information for the design of a full scale plant. It was hence decided to start pilot plant experiments.

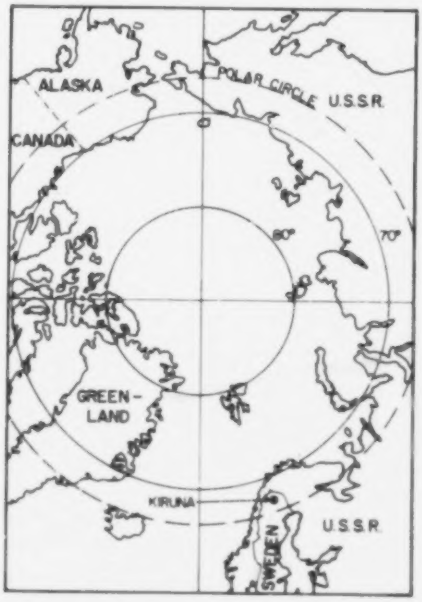

FIGURE 1 Location of Kiruna, Sweden

EXPERIMENTS WITH BIOLOGICAL TREATMENT

Activated Sludge Experiments

Apparatus

The pilot plant consisted of three aeration tanks mounted in series wtih working volumes of 2.0 m^3 (71 cu ft), 2.0 m^3 (71 cu ft), and 1.7 m^3 (61 cu ft). The settling tank was conical with a volume of 2.7 m^3 (96 cu ft) and a surface area of 2.5 m^2 (270 sq ft). The tanks were provided with inserted walls so that back-mixing was minimized. Air was supplied through perforated plastic piping on the bottoms of the tanks. Sampling (24-hour composites) was done with automatic samplers.

Sewage

The experimental unit was supplied with presettled sewage from the existing plant in Kiruna. The Kiruna sewage is almost entirely of domestic origin. The strength of the sewage is, however, quite low.

Experiments

During June, 1963 to May, 1964, 6 runs were made. The details are extensively described in duplicated reports (Balmér, Berglund and Granstrand, 1964; Balmér, Berglund and Widell, 1964) (in Swedish) and summarized by Gustavsson and Westberg (1965).

The operating conditions during the runs are given in Table 1, and detention times and calculated load factors are shown in Table 2. In all runs except run 3 the unit was run as a conventional activated sludge process. In run 3 the unit was run according to the contact stabilization principle with step addition of return sludge (Balmér, Berglund and Enebo, 1967). Flow-sheets for the runs are given in Figure 2.

TABLE 1

Mean Operating Conditions

Period	Length of run days	Temp. °C	Sewage flow m^3/hr	Recirculation ratio	Air flow m^3/hr	Suspended solids in aerator ppm	Oxygen concentration tank 1 ppm	tank 2 ppm	tank 3 ppm
1	33	9.4	1.47	0.42	14	3,000	0.4	0.8	1.0
2	34	7.8	1.36	0.42	28	3,600	0.7	1.0	1.4
3*	17	6.3	1.21	0.45	31	3,300/4,200	0.0	0.0	0.4
4	19	5.8	0.96	0.42	24	3,600	0.9	1.1	2.7
5	11	3.7	1.19	0.34	26	3,200	4.4	4.0	6.9
6	24	5.4	1.80	0.33	21	2,300	2.3	2.3	2.3

*Step addition of return sludge. Suspended solids concentration in return sludge aeration tank 12,400 ppm.

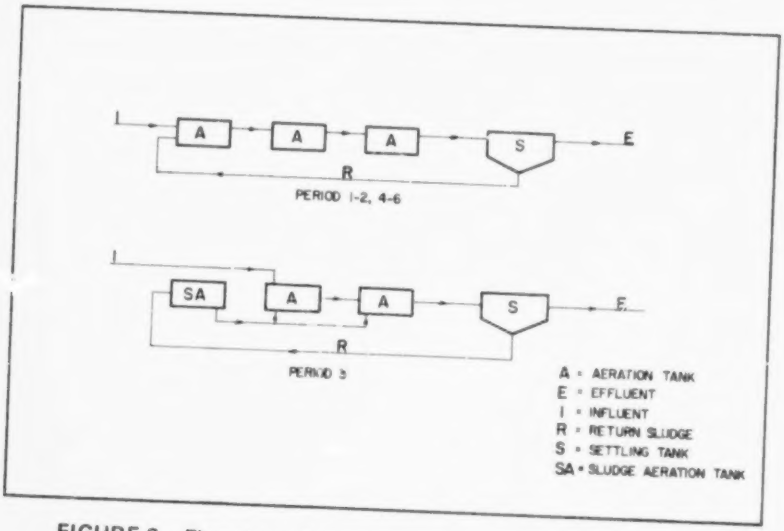

FIGURE 2 Flow sheet of experimental runs (June 1963-May 1964)

Results

Mean results from the 6 runs are presented in Table 3. In period 1 steady state conditions were probably not reached.

The oxygen content in the aeration tanks was very low throughout periods 1 - 3, indicating that oxygenation capacity of the equipment was the limiting factor. In periods 4 - 6 the oxygen content in the tanks was higher although the oxygen supply was lower compared to periods 2 and 3. It is therefore probable that bacterial activity was the rate determining factor in these periods. The mean temperature difference between periods 3 and 4 is small and the difference in bacterial activity is probably not as large as the differences in oxygen concentration in the aeration tanks may indicate. The low oxygen concentration during period 3 (contact stabilization) is explained by the large amount of suspended solids in the reaeration tank (tank 1) and the rather high load on the two aeration tanks.

The conclusions of the activated sludge experiments in pilot plant scale were:

> Biological treatment is possible at temperatures as low as 3° - 4° C (37° - 39° F). The activity of the activated sludge is seriously affected by low temperatures. Long aeration periods are necessary. If a low BOD sewage is treated, a period of at least 4 hours is required to reach an effluent BOD of 20 - 25 ppm.
>
> The contact stabilization technique is attractive as it gives a larger amount of sludge actively metabolizing in a given aeration volume.

As biological treatment now was proved feasible, the town planned to build a treatment plant

TABLE 2

Load Factors

Period	Detention time aeration hr	Detention time settling hr	Surface load settling m^3/m^2, hr	BOD-load kg BOD/m^3, d	Sludge load kg BOD/kg sludge, d
1	3.9	1.8	0.59	0.34	0.11
2	4.2	2.0	0.54	0.46	0.13
3	3.1*	2.2	0.48	0.41	0.06
4	5.9	2.8	0.38	0.40	0.11
5	4.8	2.3	0.48	0.40	0.13
6	3.2	1.5	0.72	0.41	0.25

*Detention time in aeration tanks only. Detention time in return sludge aeration tank was 3.7 hr.

consisting of presettling and secondary treatment with activated sludge. The primary treatment unit was first erected and was put into operations in 1967.

EXPERIMENTS WITH CHEMICAL TREATMENT

In the years since the activated sludge experiments, there has been a complete turnover in the opinion on the value of biological treatment. Biological treatment is now considered of very limited value as the initial BOD load caused by domestic effluents in the receiving waters is considered low in relation to the secondary pollution caused by nutrients, mainly phosphorous compounds.

It was then discussed whether the planned addition of aeration tanks and secondary settling tanks could be replaced advantageously with a coarse presettling basin and a flocculation basin placed in front of the existing settling tanks, as illustrated in Figure 3.

TABLE 3

Analytical Results - Activated Sludge Treatment

	INFLUENT			EFFLUENT			
Period	BOD_5 ppm	Suspended solids ppm	Setteable solids ml/l	BOD_5 ppm	Suspended solids ppm	Setteable solids ml/l	No. of composite samples
1	55	46	1.7	13	29	0.5	10
2	81	84	1.0	15	14	0.1	12
3	80	56	0.7	10	16	0.1	4
4	83	54	1.0	20	18	0.1	9
5	82	54	1.1	20	21	0.1	6
6	76	51	0.4	24	7	0.1	9

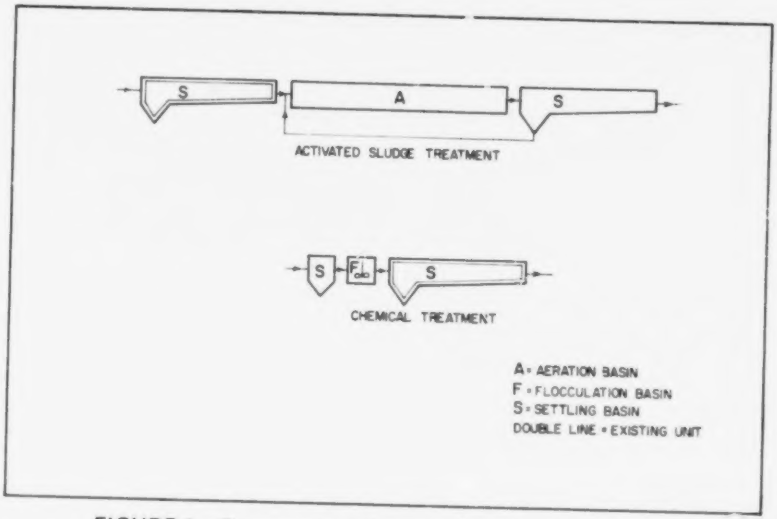

FIGURE 3 Two ways to enlarge a primary treatment plant

As a preliminary economic analysis also indicated that chemical treatment could be cheaper, this seemed to be an attractive alternative. In order to determine the treatment efficiency of a chemical treatment process, the town had to carry out pilot plant experiments.

Experiments with Flocculation and Settling

Apparatus

The pilot plant equipment (Fig. 4) consisted of a flocculation tank and a settling tank. The flocculation tank was divided into 4 compartments, each with a working volume of 0.23 m^3 (8 cu ft). The compartments were provided with paddle-type mixers with peripheral speeds of 0.5, 0.35, 0.25, 0.15 m/s (1.6, 1.2, 0.8, 0.5 ft/s) respecitvely. The settling tank was rectangular with a volume of 2.0 m^3 (70 cu ft) and a surface of 2.0 m^2 (220 sq ft).

The flocculated sewage was transferred from the flocculation tank to the settling tank through a siphon with a diameter of 10 cm (4 in). Sampling (24-hour composites) was done with automatic samplers.

The equipment proved suitable for its purpose although the peripheral speeds of the mixers in the 2 last compartments was somewhat too fast. The transfer from the flocculation tank to the settling tank also destroyed some flocs.

Sewage

The pilot plant equipment was fed with domestic sewage that was pumped from the inlet zone of one of the primary settling basins in the new treatment plant. In this way a coarse presettling was

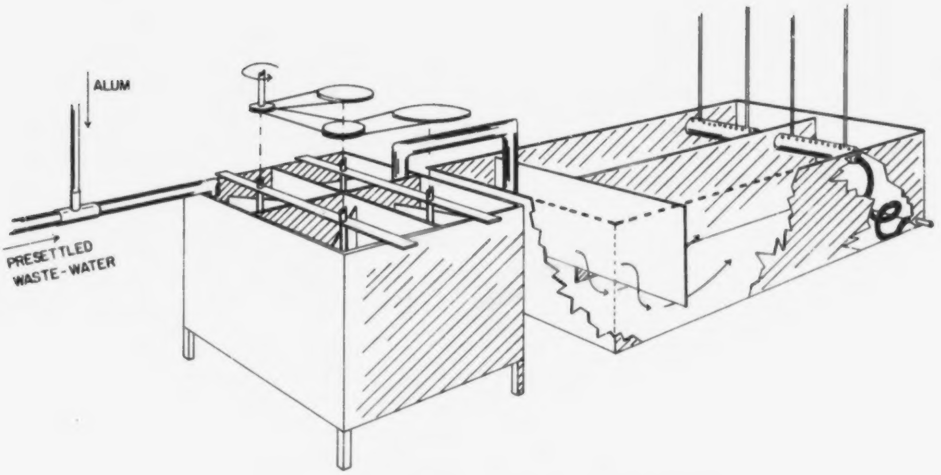

FIGURE 4 Pilot plant for chemical treatment

simulated. The sewage was almost entirely of domestic origin and the temperature of the sewage was about 3° C (37° F).

Experiments

In 4 experiments different amounts of technical grade alum (8.1% Al) were added to the sewage. In one of the runs, addition of an anionic polymer (Dow Purifloc A-23) was tried.

The flow through the apparatus was 0.96 m^3/hr (4.2 gpm) which means a detention time in the flocculation tank of about 1 hr, and of 2 hr in the settling tank.

Results

Mean results from the test runs are given in Table 4.

Experiments with Simplified Chemical Treatment

The Treatment Plant

The sewage treatment plant in Kiruna consists of a coarse bar screen, an aerated grit chamber, a comminutor and three parallel settling basins. Each settling basin has a width of 6 m (20 ft) and a length of 47 m (148 ft) and has a total volume of 640 m^3 (22,800 cu ft). During the test week the flow through the plant was almost the same every day. Flow and load factors of the settling basins are given in Figure 5.

TABLE 4

Analytical Results - Chemical Treatment

		INFLUENT			EFFLUENT			FILTERED EFFLUENT		
Alum dosage ppm	Flocculation pH	BOD_7 ppm	COD ppm	Phosphorus ppm	BOD_7 ppm	COD ppm	Phosphorus ppm	BOD_7 ppm	Phosphorus ppm	No. of Composite samples
100	6.6-6.9	60	24	6.5	41	18	2.3	28	0.20	4
120	6.4-6.9	50	25	2.9	27	15	0.62	24	0.39	3
120*	6.4-6.9	52	30	4.3	26	19	0.64	20	0.35	3
150	5.2-5.5	81	33	3.9	33	19	0.79	26	0.10	3

*1 ppm Dow Purifloc A-23 added. COD was determined with potassium-permanganate oxidation.

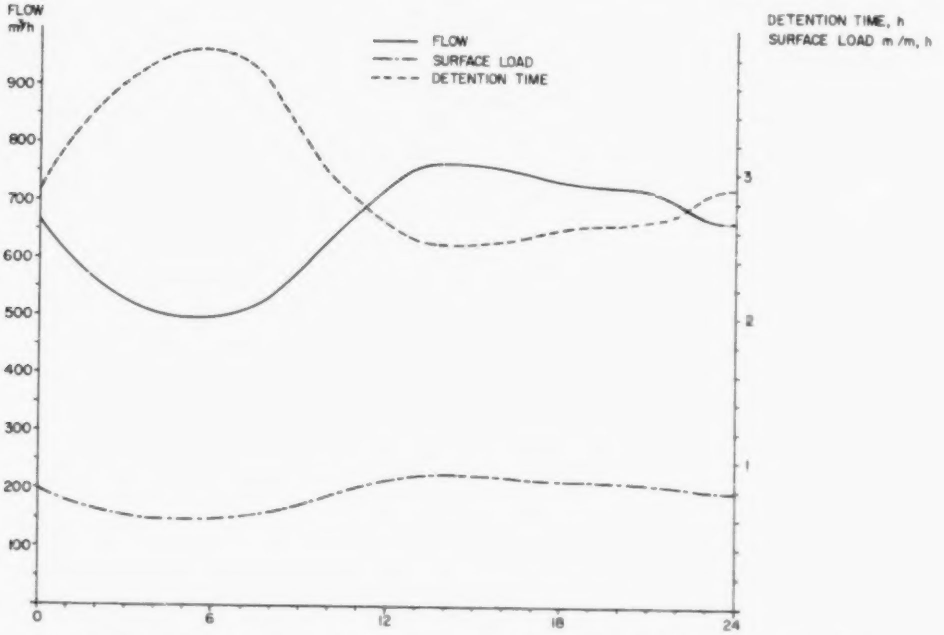

FIGURE 5 Wastewater flow and detention time and surface load on settling basins

TABLE 5

Operating Conditions - Simplified Chemical Treatment

Date	Time	Alum dosage ppm	Flocculation-pH
12/1	1100-2200	141	6.0-6.4
13/1	1300-2200	142	5.8-6.1
14/1	0900-1700	122	6.6-6.8
15/1	0800-1600	117	6.0-6.4
15-16/1	2200-1300	90	5.7-6.2
17/1	0500-1000	100	6.0-6.3
17/1	1000-1600	140	6.4-6.5

Experiments

Technical grade alum was added as a 20% solution to the influent sewage just ahead the grit chamber. The flow of alum was adjusted over the day so that a constant dose of alum per unit volume of sewage was achieved. The operating conditions are presented in Table 5.

Results

Results of the full-scale experiments are presented in Table 6 and Figure 6.

If the effluent quality with the biological treatment (Table 3) is compared to the results achieved with the chemical treatment (Table 4, Table 6 and Figure 6), it appears that the chemical treatment gives almost as low BOD values as the biological treatment. Furthermore, the chemical treatment removes up to 90% of the phosphorus content of the sewage.

The results of the full-scale experiment indicate that separate flocculation basins may be omitted.

Visual observations during the experiments revealed that the flocculation pH should be controlled, as pH over 6.7 yielded a floc with poor settling properties.

Sludge Handling

In Kiruna the normal amount of primary sludge is 50-70 g total solids per capita per day. When alum is added to the sewage, this amount increases to about 100 g total solids per capita per day. During the full-scale experiment the total solids concentration of the sludge fell from the normal 5-8% down to 2-3%. The sludge volume hence increased to about 4 times the normal.

The sludge dewatering equipment in Kiruna consists of a "roto-plug" gravity dewatering drum followed by a screw press. The equipment captures 80-90% of the total solids in the primary sludge. With chemical treatment sludges the solids capture was down to about 30%. Cationic

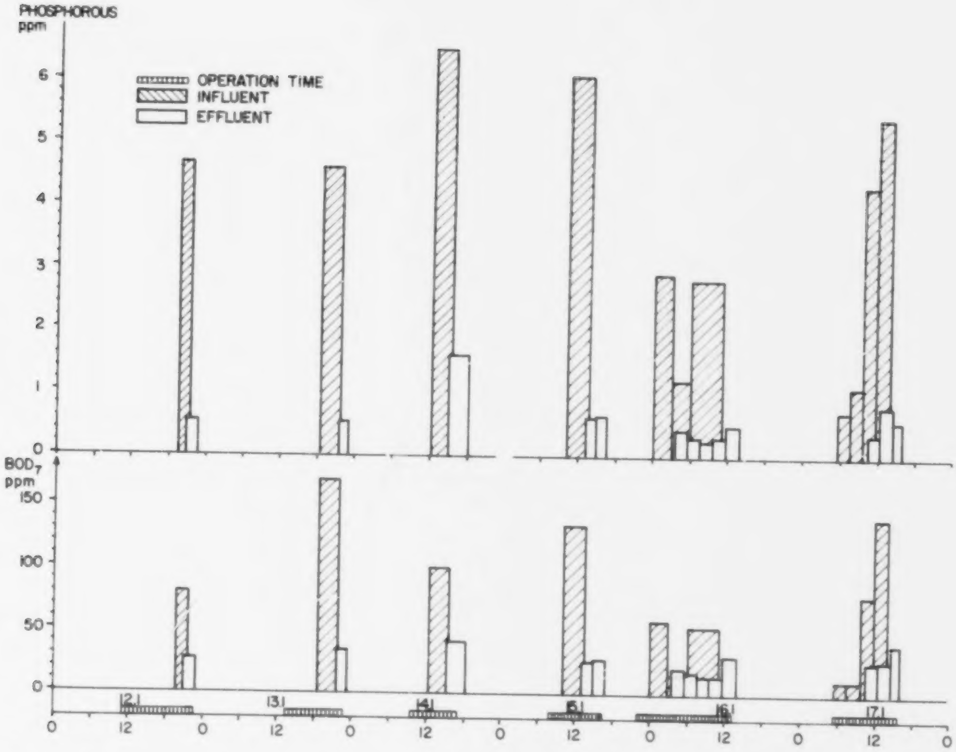

FIGURE 6 Operation time with chemical treatment and phosphorus and BOD$_7$ in influent and effluent

polymers had a very good conditioning effect on the sludge, but the dynamic action of the sludge dewatering equipment destroyed the flocs formed by the polymers. It is, however, believed that the sludge could be successfully dewatered with static-type dewatering equipment.

CONCLUDING DISCUSSION

The experiences with the activated sludge treatment have proved that biological treatment is possible even at very low sewage temperatures. As the metabolizing activity of the activated sludge bacteria is considerably reduced, long aeration periods, 4-5 hours, and therefore large aeration basins, are required.

A chemical treatment is much less sensitive to low temperatures and requires only about 0.5 hour detention time in the flocculation tanks. The difference in investment costs will in many instances be so large that the increased running costs are justified.

If a community has an existing primary treatment plant with a long detention time (more than 2 hours), it may be possible to achieve a substantial increase in treatment efficiency just by a simple addition of flocculating chemicals to the influent.

The BOD removal with chemical treatment is somewhat inferior to what can be achieved with biological treatment. This drawback, however, is compensated by the superior phosphorus removal.

TABLE 6

Analytical Results - Simplified Chemical Treatment

	INFLUENT			EFFLUENT			
Date	Time	Suspended solids ppm	COD ppm	Time	Suspended solids ppm	COD ppm	Aluminum ppm
12/1	1930-2130	120	33	2100-2245	24	14	1.3
13/1	1840-2140	110	35	2030-2230	14	13	1.3
14/1	1220-1530	130	35	1500-1820	40	16	
15/1	1000-1400	110	40	1300-1500	13	11	1.6
				1500-1715	17	11	1.1
16/1	0000-0300	60	19	0320-0520	26	15	0.8
	0300-0600	15	7	0520-0720	20	9	1.4
				0720-0920	17	18	1.3
	0600-1100	52	19	0920-1120	33	9	0.8
				1120-1400	20	17	1.0
17/1	0530-0730	4	6				
	0730-0930	7	7	1030-1230	8	8	0.8
	0930-1130	59	30	1230-1430	16	15	1.1
	1130-1330	98	42	1430-1530	16	18	1.3

*COD was determined with potassium-permanganate oxidation

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BIOLOGICAL SEWAGE TREATMENT IN A COLD CLIMATE AREA

Shigeo Terashima, Keichi Koyama, and Yasumoto Magara

INTRODUCTION

Recently many sewage treatment plants using activated sludge processes have been installed in the Hokkaido and Tohoku districts which are cold climate areas in Japan. In these plants it is necessary to give attention to low sewage temperature in winter and thaw seasons.

The rate of run-off is often about twice dry weather flow. Thawing flow carries the turbid substances that were piled up in winter and lowers the sewage temperature. Therefore, it is considered that thawing flow affects sewage treatment process, but no reports about the effect of thawing are available.

The removal characteristics of BOD and nitrogen at the Souseigawa municipal sewage treatment plant (Sapporo, Japan) from March 1967 to February 1970, are shown in Table 1.

TABLE 1

The Effect of Temperature on Purification at the Souseigawa Plant

Sewage temp.	BOD removal (%)	(Org-N. + NH_3-N.) removal (%)
<10° C	82	0
11-15° C	82	23
>16° C	87	25

This shows that nitrogen removal ceases below 10° C but BOD removal is little affected by sewage temperature. It is assumed that there is a different effect on BOD and nitrogen removal activity of activated sludge.

The activated sludge process is composed of (1) removal of wastes, mainly by biosorption, after contact of the waste with activated sludge, (2) removal of wastes by the anabolism and catabolism of the absorbed substrate, (3) aeration to supply the oxygen for these reactions and, (4) solid-liquid separation. Several reports have been published on the effect of temperature on activated sludge processes (Banerji, 1965). Ludzack et al. (1952) showed that sewage temperature had little effect on the activated sludge. The waste removal at 30° C was 10% higher than at 5° C, but the solids yield was greater at 5° C than 30° C. (Keefer, 1962) reported operating results over a 20-year period for a municipal sewage treatment plant. The variation of the average percent BOD removal with temperature was dependent on the flow and increased as the organic load was increased. Okun (1949) reported little effect of temperature on BOD removal in the activated sludge process between 8-35° C.

These reports discussed the overall effects of temperature on the activated sludge process and were not investigations of each individual process component of the total system. There have been no investigations on compounds other than BOD.

It is necessary to investigate the effect of each process and to know what process is mainly affected by sewage temperature. It is possible to have a good activated sludge plant in a cold climate area by strengthening that process mainly affected by sewage temperature.

We have been investigating the thawing effect on sewage and the effect of low sewage temperature on the activated sludge process. This paper deals with the effects of thawing and the effects of low sewage temperature on the aeration process, substrate removal, nitrification and settling characteristics of activated sludge.

THE EFFECTS OF THAWING ON SEWAGE

Procedures

Thawing is affected by atmospheric temperature, sunshine, rainfall, underground temperature, etc. Atmospheric temperature is considered the primary influence for run-off analyses. Degree-days and degree-hours are used as indices of atmospheric temperature. The degree-day is defined as the cumulative number of daily mean atmospheric temperatures above 0° C within a certain period, and degree-hour as the cumulative number of hourly atmospheric temperatures above 0° C within the day. Sewage drainage areas are more affected by atmospheric temperature than river catchment areas because sewage drainage areas are very limited and thawing is directly related to hourly change of atmospheric temperature. So, the degree-hour is used as an index of thawing run-off analysis.

We investigated two sewage drainage districts at Sapporo, Japan. Their characteristics are shown in Table 2.

TABLE 2

Characteristics of Drainage Districts

	Souseigawa District	Makomanai District
Drainage Area	623 ha (April 1968) 736 ha (April 1969)	37 ha (April 1968)
Population	125,000 (Dec. 1967)	
Sewer System	Combined system	Separate system
Type of District	Urban area	Residential area

The rate and quality of sewage flow were observed at the receiving well of the sewage treatment plant (1968) or at the final pumping station (1969) at the Souseigawa district; the same were examined at the outlet of the storm sewer at the Makomanai district. Sewage was analyzed for temperature, pH, 4.3 alkalinity, chloride ion (Cl^-), total residue (T-Re), suspended solids (SS), chemical oxygen demand (COD), biological oxygen demand (BOD), organic nitrogen (Org N), ammonium nitrogen (NH_3-N), nitrite nitrogen (NO_2-N), and nitrate nitrogen (NO_3-N) according to the 12th edition of **Standard Methods** (1965). Sampling was done at 3:30 PM - 4:00 PM (Souseigawa district) and 2:30 PM - 3:00 PM (Makomanai district). In Sapporo, it was most probably to find the highest atmospheric temperature at about 2:00 PM. Therefore it was considered that this sample was most affected by thaw. To calculate the degree-hour, an atmospheric temperature was summed from 3:00 AM to the next day at 3:00 AM. The rate of thawing run-off was gained by reducing the dry weather flow (55,000 M^3/d-1968, 60,000

M^3/d-1969) from the total rate of sewage flow. If there was a rainfall, the rate of storm run-off was reduced.

Results and Discussion

The Rate of Thawing Run-off

Figures 1, 2 and 3 show the daily aspects of a degree-hour and the rate of thawing run-off. The relationships between a degree-hour and the rate of thawing run-off are shown in Figure 4. The next empirical formulas (1-1), (1-2), and (1-3) are attained from this relationship.

Souseigawa district

March 1 - April 6, 1968

$$Q_t = 0.743\,A\,D + 17.2A \qquad (1\text{-}1)$$

correlation coefficient r = 0.87

March 6 - March 26, 1969 (excluding March 21)

$$Q_t = 0.747\,A\,D + 27.7A \qquad (1\text{-}2)$$

correlation coefficient r = 0.94

Makomanai district

1969 (duration of cold days)

$$Q_t = 0.14\,A\,D + 9.7A \qquad (1\text{-}3)$$

correlation coefficient r = 0.94

Here, Q_t = the rate of thawing run-off (M^3/D)
D = the degree-hour (° C hr.)
A = drainage area (ha)

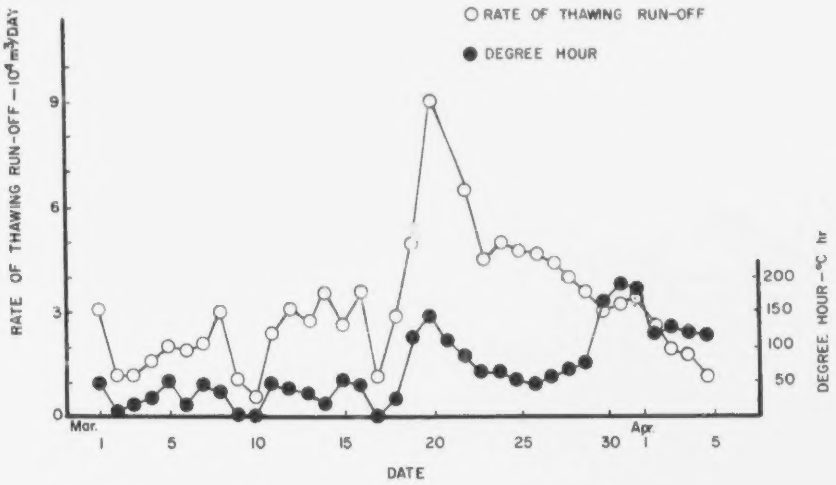


FIGURE 1 Rate of thawing runoff and degree-hour (Souseigawa district, 1968)

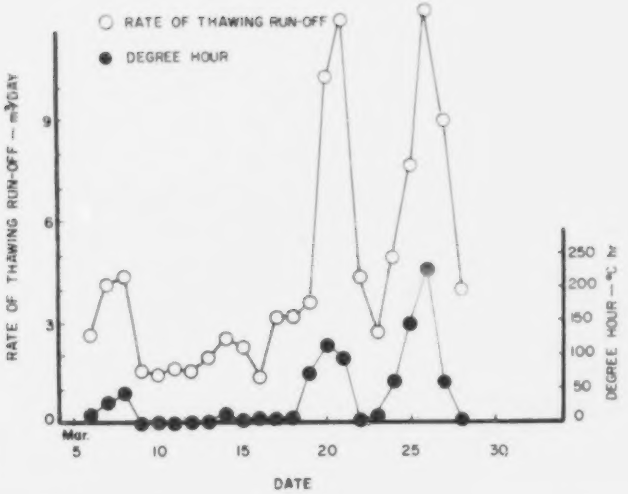


FIGURE 2 Rate of thawing runoff and degree-hour (Souseigawa district, 1968)

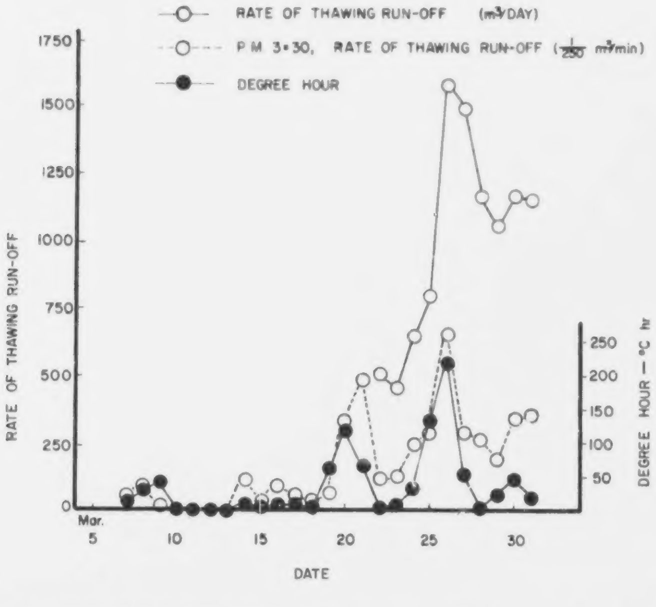


FIGURE 3 Rate of thawing runoff and degree-hour (Makomanai district, 1969)

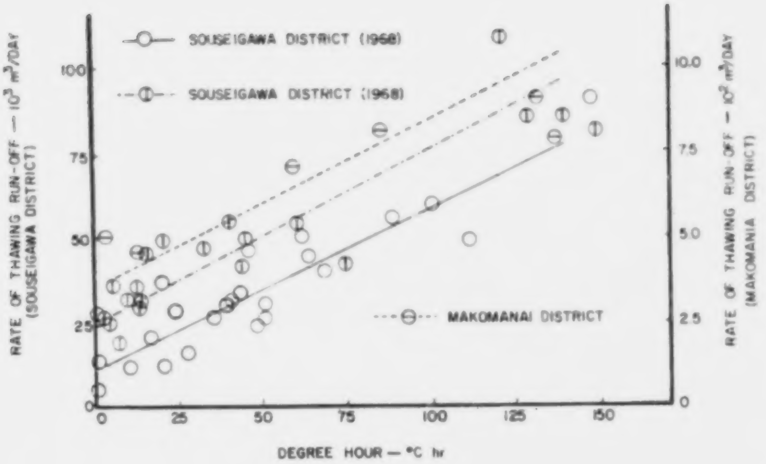


FIGURE 4 Correlated runoff and degree-hour (Souseigawa and Makomanai districts)

The result shows there is significant correlation during the limited periods when the ground was covered by snow and the ground temperature was low. This correlation did not become obvious on a rainy day because thawing was accelerated by storm run-off. And, during the close of the thawing season (after March 29, 1968, April 7, 1969), there was not significant correlation.

The rate of thawing run-off per unit drainage area of the Makomanai district was smaller than that of the Souseigawa district. It was assumed that this was a characteristic of the drainage district. The sewer density (total length of sewer per unit area) and imperviousness in the Makomanai district were relatively smaller than those of the Souseigawa district, and the surface running water was able to flow into domestic sewers through manholes, decreasing the rate of run-off in storm sewers from actual thawing run-off.

The degree-hour was calculated from hourly thermometer records. There is a method to determine the degree-hour from the highest and lowest temperature of the day (Sakai, 1963). Figure 5 shows that the calculated values agree very well with the observed values. If it is possible to obtain a highest and a lowest temperature of the day from the meteorological observatory, then one can estimate the rate of thawing run-off.

Sewage Quality

Sewage temperature in winter, excluding the thawing season, is about 9° C at the Souseigawa district and about 6° C at the Makomanai district. However, sewage temperature decreased to 4.5° C (March 21, 1969, Souseigawa district) and 2° C (March 15, 1969, Makomanai district) by inflow of run-off waters during a thaw.

The correlation between a sewage temperature and the percentage of thawing run-off during dry weather flow at Souseigawa is shown in Figure 6. As shown in this figure, sewage temperature is

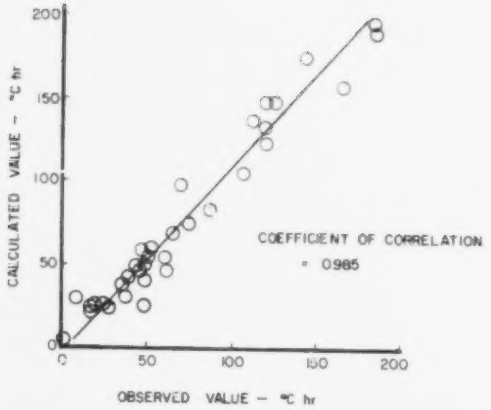


FIGURE 5 Correlation between observed and calculated degree-hour

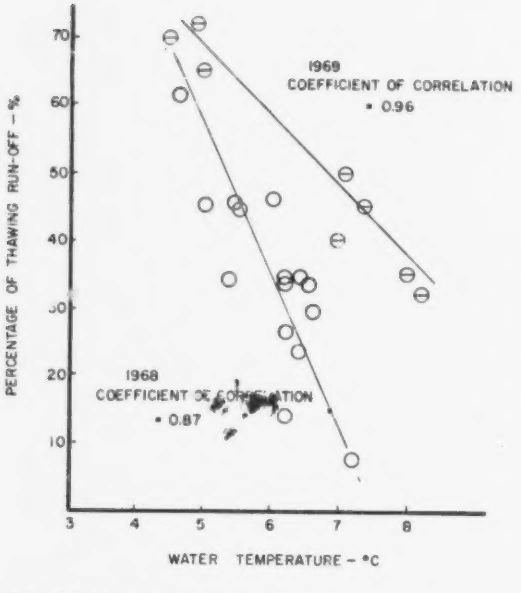


FIGURE 6 Correlation between water temperature and runoff percentage

very significantly affected by thawing inflow. However, at the Makomanai district, there was poor correlation compared to the Souseigawa district during the season. Hourly observation data (Figure 7) shows the reason why significant correlation was not attained. The correlation at the end of the season differed from that at the beginning. This was caused by changing sewage temperature, influenced mainly by ground temperatures.

Suspended matter was increased up to 1,500 mg/l at the Makomanai district and up to 500-600 mg/l at the Souseigawa district by thawing. The correlation of suspended solids and percentage of thawing run-off at the Souseigawa district is shown in Figure 8-1. The correlation of suspended solids and run-off rate at the Makomanai district is shown in Figure 8-2. Each figure shows that suspended solids and chemical oxygen demand attributed to suspended solids increase accordingly with increments of thawing run-off. The results of the Makomanai district show that thawing run-off in early periods contains much turbidity compared with final periods.

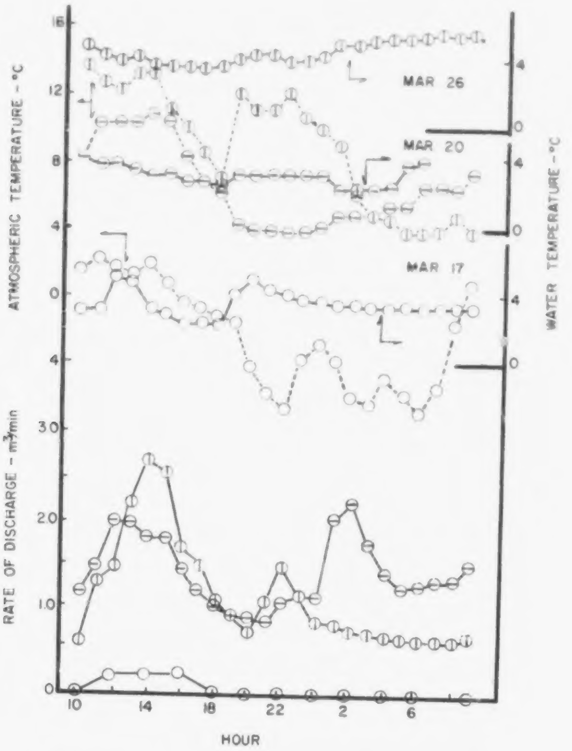


FIGURE 7 Hourly observation at Makomanai

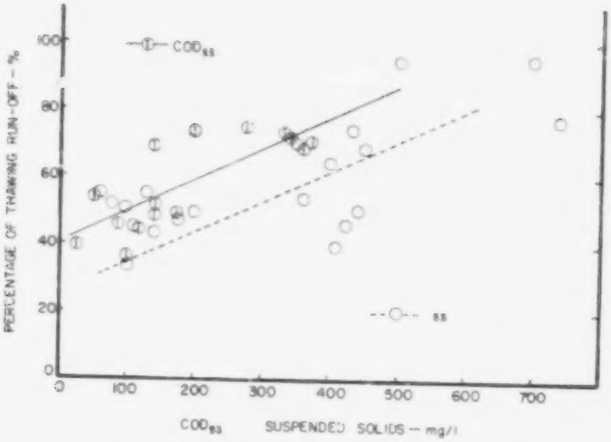


FIGURE 8-1 Correlation among percentage of runoff, COD$_{ss}$ and suspended solids (Souseigawa district, 1969)

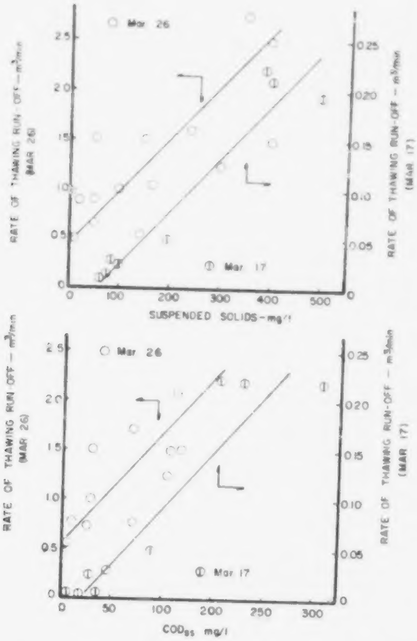


FIGURE 8-2 Correlation among rate of runoff and suspended solids, COD$_{ss}$ (Makomanai district)

Other components of the sewage are shown in Figure 9. The figures show that generally each component, except chloride and nitrate, increases with temperature decrement which relates to the rate of thawing run-off. However, the range of those fluctuations was not so large as to affect the sewage treatment facilities.

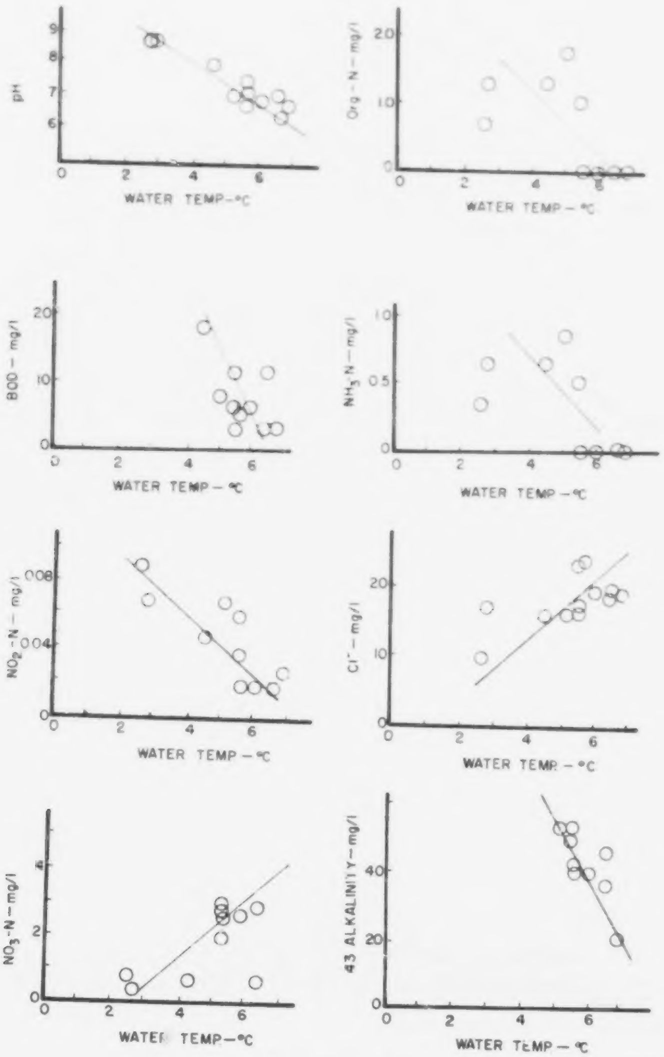


FIGURE 9 Various sewage components

Conclusion

The rate of thawing run-off significantly relates to the degree-hour. If these correlations are known for a certain drainage area, it is possible to predict the rate of thawing run-off from the highest and lowest atmospheric temperatures of the day. However, these correlations become less obvious at the end of the thaw season.

Sewage temperature and suspended solids are significantly affected by thawing run-off, especially when sewage temperature drops below 5° C.

THE EFFECT OF LOW SEWAGE TEMPERATURE ON THE ACTIVATED SLUDGE PROCESS

The Effect of Sewage Temperature On Oxygen Transfer

Procedures

The effects of temperature on oxygen transfer in a diffused aeration tank were investigated by measuring the oxygen transfer coefficient at every 5° C decrement from 25° C to 5° C under a fixed rate of air flow.

Completely mixed aeration tanks were used, with porous plate diffusers installed in the bottom. The aeration tank is shown in Figure 10. The oxygen transfer coefficient was obtained by a non-steady state procedure (Eckenfelder, O'Connor, 1961) under stabilized operating conditions. The aeration tank was filled to a desired volume with tap water. Dissolved oxygen in the water was removed by admitting nitrogen gas. After the dissolved oxygen approached 1 mg/l, air was admitted at 10 l/min or 20 l/min, and the dissolved oxygen was continuously measured by an oxygen analyzer (Toshiba-Beckman Co. Ltd., 777 type).

Oxygen transfer can be expressed by Formula 2, using an overall oxygen transfer coefficient.

$$\frac{dC}{dt} = K_{La}(Cs\text{-}C) \qquad (2)$$

Here, K_{La} = overall oxygen transfer coefficient
C = dissolved oxygen concentration
Cs = saturated dissolved oxygen concentration

The transfer coefficients computed from the above relationship are shown in Table 3 and Figure 11. These results show that the oxygen transfer coefficient is affected by temperature. This effect can be expressed by Formula 3:

$$K_{La}T = K_{La\,20} \times \theta^{(T\text{-}20)} \qquad (3)$$

in which

T = temperature (° C)

$K_{La}T$ = transfer coefficient at temperature T

$K_{La\,20}$ = transfer coefficient at 20° C

θ = temperature coefficient

The temperature coefficient, which is computed from Formula (3), was found to be 1.017. This value agreed favorably with the result by Eckenfelder (1961).

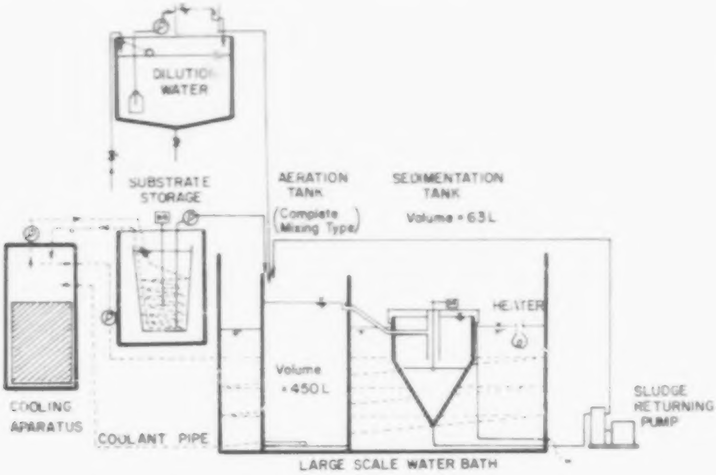

DILUTION WATER
SUBSTRATE STORAGE
AERATION TANK
(Complete Mixing Type)
SEDIMENTATION TANK
Volume = 63 L
HEATER
Volume = 450 L
COOLING APARATUS
COULANT PIPE
LARGE SCALE WATER BATH
SLUDGE RETURNING PUMP

FIGURE 10 Complete mixing continuous flow units

TABLE 3

The Effect of Temperature on Oxygen Transfer Coefficient

Water Temperature	Oxygen Transfer Coefficient (1/hour) 10 l/min	20 l/min
5° C	2.6	5.2
10° C	3.0	6.5
15° C	3.2	6.1
20° C	3.5	6.9
25° C	3.2	7.4

The ratio of the mass of oxygen transfer at any water temperature to the transfer at 20° C is calculated from Formulas 2 and 3, assuming the dissolved oxygen concentration is to be held constant, for instance, at 2 mg/l. This value, shows, as in Figure 12, that the mass of oxygen transferred increases with water temperature decrement. The mass of oxygen transfer is more affected by increasing a saturated dissolved oxygen concentration than by decreasing an overall oxygen transfer coefficient. From this result, there need be no concern that oxygen transfer will be hampered by a lowering of the water temperature.

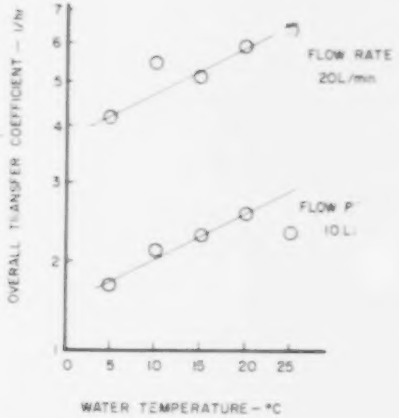


FIGURE 11 Oxygen transfer coefficient and temperature

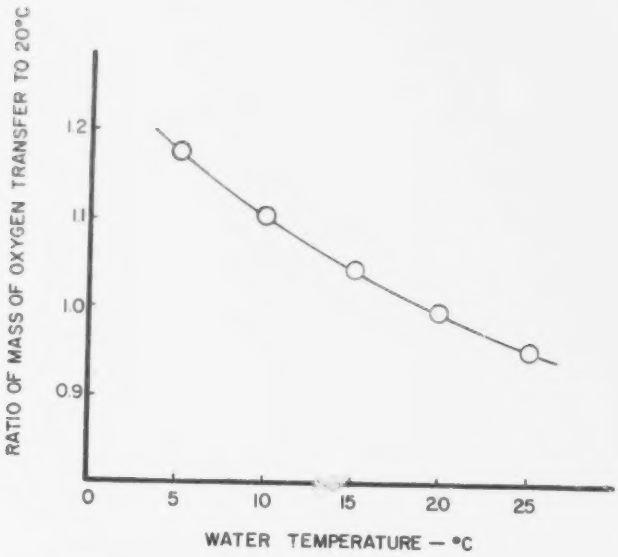


FIGURE 12 Ratio of mass of oxygen transfer

The Effects of Temperature On the Purification Activity of Activated Sludge

Procedure

Experiments shown in Table 4 were performed to find the effect of temperature on the activated sludge process.

TABLE 4

Experimental Details

Run No.	Substrate	Aeration tank	Aeration period	Water temp.	Operation
1-1	sewage	M.C.A.T.[1]	6 hr	20° C	C.F.,[4] 1440 l/D
1-2	"	"	"	15° C	"
1-3	"	"	"	10° C	"
2-1	skim milk	C.M.A.T.[2]	"	20° C	"
2-2	"	"	"	"	C.F., 1080 l/D
2-3	"	"	"	15° C	C.F., 1290 l/D
2-4	"	"	"	10° C	C.F., 704 l/D
3-1	skim milk + peptone	M.C.A.T.[3]	"	20° C	C.F., 1440 l/D
3-2	"	"	"	15° C	"
3-3	"	"	"	10° C	"
3-4	"	"	"	5° C	"
4-1	skim milk			20° C	Batch
4-2	"			10° C	"
5-1	glucose			20° C	"
5-2	"			15° C	"
5-3	"			10° C	"

(1) multi-cell (6 cells) aeration tank
(2) completely mixed aeration tank
(3) multi-cell (14 cells) aeration tank
(4) continuous flow

Apparatus. The apparatus used in the experiments is shown in Figure 13. A plug flow aeration tank was used in runs No. 1 and No. 3. This aeration tank was simulated from an actual aeration tank of a sewage treatment plant by dividing it into multi-cells.

Substrate. The model plant (run No. 1), which was fed domestic sewage, was installed in the Makomanai sewage treatment plant. Preclarified sewage was used as a substrate, and the model plant fed under load fluctuation, which caused only fluctuation of sewage concentration. Synthetic sewage (runs No. 2 and No. 3) had ammonium chloride added to supply nitrogen to the activated sludge.

Operation. Model plants were normally operated at 1 l/min. inflow and 0.25 l/min. return sludge. Under these loadings, aeration periods were 6 hours and sedimentation periods 1 hour. However, when suspended solids were carried over from the sedimentation tank, and the aeration tank could not sustain the desired activated sludge concentration, the rate of inflow was decreased. The volume of return sludge was then increased so that the aeration period would remain constant at 6 hours.

Acclimation was performed by increasing an organic load from low values to a set value. Domestic sewage sludge was used for seed. Later, to confirm acclimation, testing was started. Batch experiments were performed using continuous flow plant sludge.

Analysis. The sample was analyzed for water temperature, pH, dissolved oxygen, total residue, suspended solids, volatile suspended solids, chemical oxygen demand, biological oxygen demand, organic nitrogen, ammonium nitrogen, nitrite nitrogen, nitrate nitrogen and sludge volume index (SVI) according to the 12th edition of **Standard Methods** (1965).

Results and Discussion

The results of continuous flow plants are shown in Table 5 and the results of batch experiments are shown in Table 6.

Settling Characteristics. The sludge volume index (SVI) proved that sludge settling characteristics depend strongly on the sewage temperature, as in Figure 14.

TABLE 5

Results of Continuous Flow Plant

	Run No.										
	1-1	1-2	1-3	2-1	2-2	2-3	2-4	3-1	3-2	3-3	3-4
Water temp. ° C	20	15	6	20	20	15	10	20	15	10	5
Organic load[1]	0.25	0.35	0.38	0.34	0.56	0.17	0.15	0.33	0.47	0.25	0.14
MLSS[2]	3778	2995	2836	3035	866	2336	2431	2636	1000	2647	3219
SVI	90	178	181	45	-	322	402	51	112	345	250
Influent											
COD[2]	286	346	365	322	152	138	233	275	157	161	147
Org-N+NH_3-N[2]	55.9	63.8	67.3	34.4	10.0	10.4	17.4	27.5	15.7	16.1	14.7
Effluent											
COD[2]	83	45	274	51	-	9	115	52	27	26	45
CODs[3,2]	45	31	37	31	-	8	13	17	11	9	7
SS[2]	43	22	209	23	-	26	117	51	11	15	32
Org-N[2]	2.1	1.8	17.6	-	7.1	1.2	15.8	2.6	2.6	4.6	1.2
NH_3-N[2]	18.6	34.6	35.5	-	1.1	0.2	7.8	8.9	7.5	2.3	3.8
NO_2-N[2]	7.2	5.3	0	-	2.1	0.2	0	0.36	2.1	0	0
NO_3-N[2]	1.4	1.2	0	-	11.4	14.9	7.1	4.4	0.7	3.2	0
COD removal	71	87	25	85	-	93	51	81	83	83	69
%[4]	89	91	90	91	-	94	94	96	93	94	96
COD removal[5]	327	414	128	394	-	167	83	292	188	192	144
Org-N+NH_3-N removal %	63	43	22	-	18	87	0	58	29	57	53
NO_2-N+NO_3-N removal %	14	7	0	-	135	145	58	17	17	19	0
Nitrification Activity[6]	1.1×10^{-2}	0.8×10^{-2}	0	-	6.2×10^{-2}	2.5×10^{-2}	1.2×10^{-2}	0.7×10^{-2}	1.1×10^{-2}	0.6×10^{-2}	0

(1) Organic load = kg COD/kg MLSS/day
(2) Concentration = mg/l
(3) CODs = Soluble COD
(4) Calculated from CODs
(5) COD removed = g COD/day
(6) Nitrification activity = (NO_2-N+NO_3-N) MLSS day

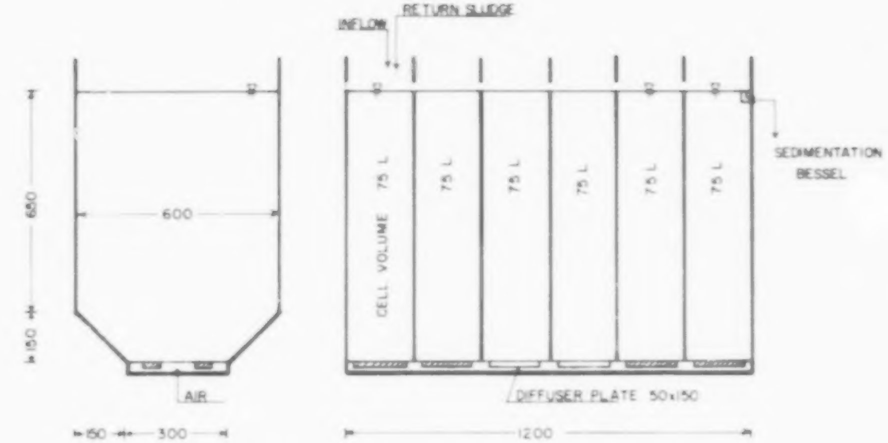

RETURN SLUDGE
INFLOW
SEDIMENTATION BESSEL
680
600
150
CELL VOLUME 75 L
75 L
75 L
75 L
75 L
75 L
AIR
DIFFUSER PLATE 50x150
150
300
1200

FIGURE 13-1 Multi-cell aeration tank (6 cells)

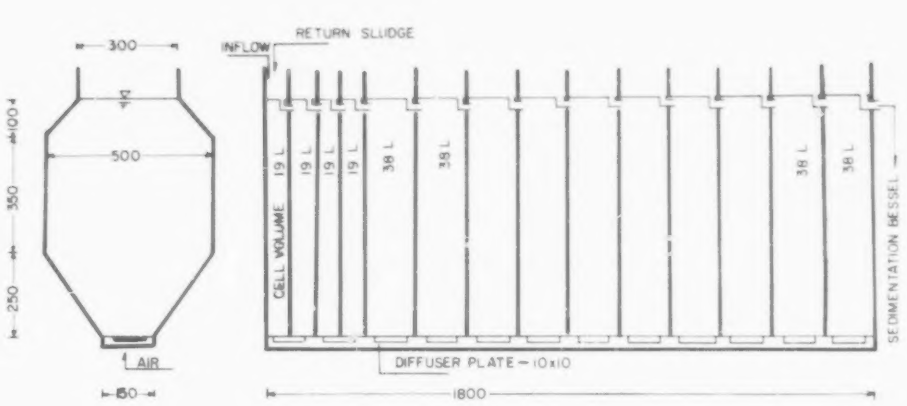

300
RETURN SLUDGE
INFLOW
100
500
350
250
CELL VOLUME 19 L
19 L
19 L
19 L
38 L
38 L
38 L
38 L
SEDIMENTATION BESSEL
AIR
DIFFUSER PLATE – 10x10
150
1800

FIGURE 13-2 Multi-cell aeration tank (14 cells)

The sedimentation of activated sludge mixed liquor is zone settling. It is said that zone settling velocity is related to the void ratio, which is proportional to the 0.225th power of Reynold's number (Tanbo, 1969). When sewage temperature is decreased from 20° C to 5° C under the same physical and biochemical properties of activated sludge floc, the SVI is estimated to be 1.1th of 20° C's value from Formula 4. This result is not the same as the experimental result.

$$SVI = \frac{1-e}{MLSS} \times 100 \qquad e = A\,Re^{0.225} \tag{4}$$

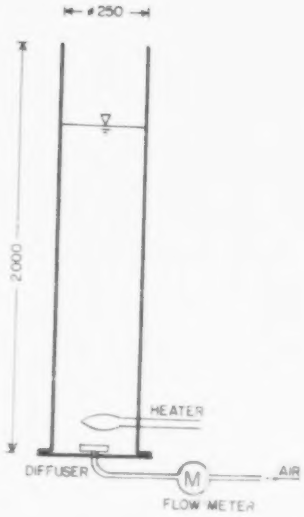


FIGURE 13-3 Batch unit

Here, e = void ratio
Re = Reynold's number
A = coefficient of flow properties

Figure 15 shows the change of SVI and substrate concentration after substrate contact with activated sludge. SVI increased in gradual proportion to absorbed substrate and showed maximum values at the time when substrate was almost removed from the solution. Afterward, SVI gradually decreased with the utilization of absorbed substrate. Therefore, it was considered that SVI was related to the absorbed substrate.

It is considered from the above description that the high value of SVI at low water temperature is not only attributable to change of water viscosity but also to changes of activated sludge floc physical properties like density, bound water ratio, and surface electric charge. Further investigations into these relationships between the biological and physical properties are necessary.

COD. The removal ratio of soluble COD was not affected, but COD that contained suspended matter was affected, by water temperature in a continuous flow aeration unit, as in Figure 16. This showed an increase of suspended matter, carried over from the sedimentation tank in low water temperatures. Total mass of COD removed decreased with water temperature, as in Figure 17. This was the result of floc carry-over at the Makomanai model plant; other model plants were operated under low organic load to repress the high SVI or low overflow rate of the sedimentation tank to prevent floc carry-over.

TABLE 6

Results of Batch Experiments

	Run No.						
	4-1		4-2		5-1	5-2	5-3
Water temp.	20° C		10° C		20° C	15° C	10° C
Organic load*	0.34	0.18	0.40	0.30	0.18	0.18	0.18
MLSS mg/l	3500	3500	3000	2000	2200	2200	2200
COD initial (mg/l)**	278	139	286	66			
COD 6 hrs (mg/l)	54	49	44	18			
K_{COD}	2.5 x10^{-2}	2.2 x10^{-2}	0.9 x10^{-2}	1.1 x10^{-2}			
$K_{Org.-N}$	6.05 x10^{-4}	6.50 x10^{-4}	4.08 x10^{-4}	3.72 x10^{-4}	8.7 x10^{-4}	3.2 x10^{-4}	2.5 x10^{-4}
$K_{Nit.}$					3.9 x10^{-2}	2.3 x10^{-2}	1.4 x10^{-2}

* Organic load = kg COD/kg MLSS/Day

** COD initial was determined after 3 min. of aeration K = l/day

The removal of COD is shown in Formula 5:

$$\frac{dL}{dt} = -KSL \tag{5}$$

Here, L = substrate concentration
S = initial activated sludge concentration
K = substrate removal rate

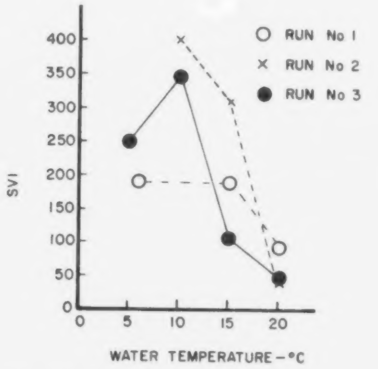


FIGURE 14 SVI and water temperature

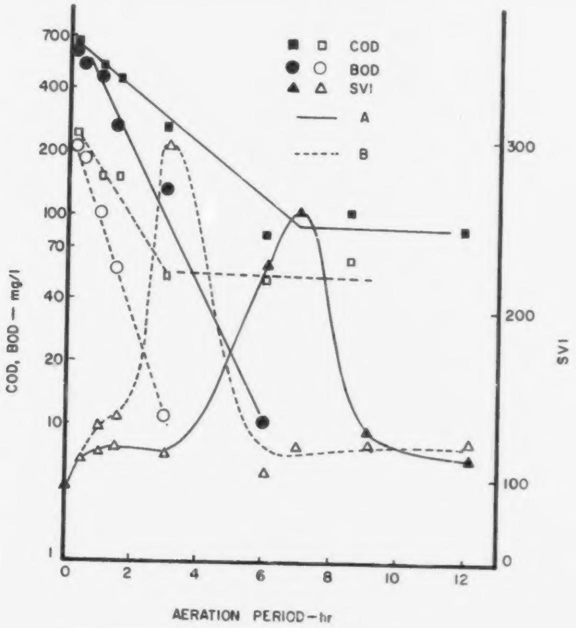
Soluble COD removal rate in batch tests (Table 6) showed it decreased with sewage temperature. Temperature coefficient (θ) which was obtained according to Formula 6 is 1.089.

$$K_T = K_{20}\theta^{(T-20)} \tag{6}$$

Here, K_T = the rate at T° C
K_{20} = the rate at 20° C
T = water temperature
θ = temperature coefficient

This (θ) showed that the removal rate was about half when the water temperature was decreased by 10° C. Figure 16 shows that the removal rate was not affected by temperature in a continuous flow aeration tank as in the batch experiments. This was due to the effluent of continuous flow, sampled after 6 hours of aeration.

There is a relationship between settling characteristics and absorbed substrate of activated sludge that is shown in Figure 15. And the removal rate is affected by absorbed substrate of return



CONDITION OF EXPERIMENT
Water temperature = 20°C Substrate = skim milk + peptone
A — Organic load = 0.50 kgCOD/kgMLSS Day MLSS = 2950
B — Organic load = 0.25 kgCOD/kgMLSS Day MLSS = 2540

FIGURE 15 Characteristics of SVI and substrate removal

sludge. From these points, it is assumed that the stabilization time for oxidation or synthesis of absorbed substrate (that is, the aeration time after the removal by absorption) is the key to successful operation of the activated sludge process.

Nitrogen. The primary nitrogen components of influence to sewage treatment plants are organic nitrogen and ammonium nitrogen. These components are used by bacteria and converted to cell nitrogen or to nitrite and nitrate nitrogen. Nitrification is very sensitive to dissolved oxygen in mixed liquor, so results in Tables 5 and 6 are from experiments performed above 3 mg/l of dissolved oxygen.

The results of batch units (Fig. 18) show that the organic and ammonium nitrogen removal are first order reactions, expressed by Formula 5, and nitrification is a zero order reaction, shown by Formula 7.

$$\frac{dL}{dt} = KS \tag{7}$$

Here, L = nitrite and nitrate nitrogen
S = mixed liquor suspended solids
K = rate of nitrification

The removal rates of organic and ammonium nitrogen and the rate of nitrification (Table 6), determined according to Formulas 5 and 7, show that these rates decrease with temperature. The temperature coefficient (θ) according to Formula 6 is 1.092 for organic and ammonium nitrogen and 1.106 for nitrification.

The experimental results of continuous flow plants in Figure 19 show the same phenomena as in municipal sewage treatment plants of the same organic load. When the temperature is decreased to 5° C, nitrification does not occur and the nitrogen compounds in the influent are discharged without stabilizing.

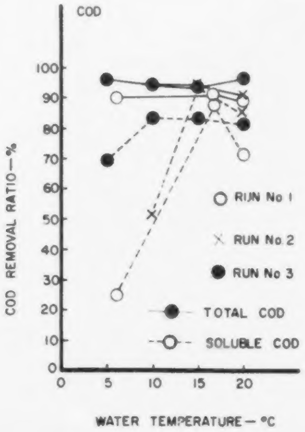


FIGURE 16 COD removal ratio and water temperature

From the above, the rate coefficient of nitrogen removal or nitrification is smaller and more sensitive to temperature than that of COD. The nitrogen removal reaction is not completed in 6 hours of aeration as is COD, and the nitrogen compounds in the influent are discharged with partial or no stabilization from the aeration plant.

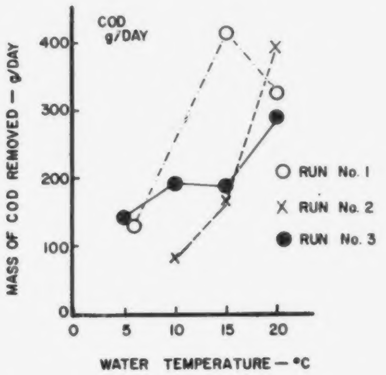


FIGURE 17 Mass of COD removed and water temperature

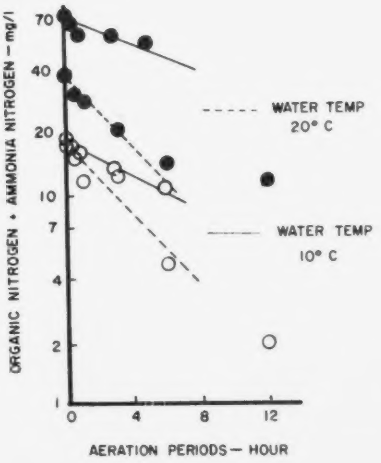


FIGURE 18-1 Effect of water temperature on nitrogen decrement

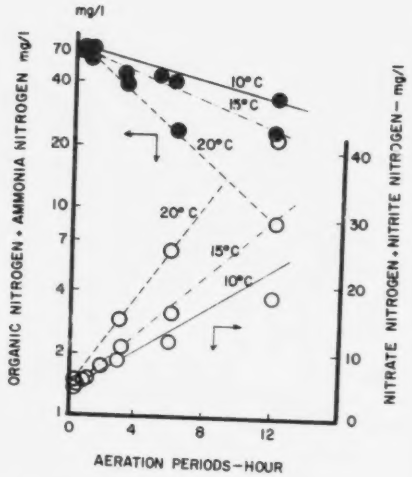


FIGURE 18-2 Effect of water temperature on nitrogen removal

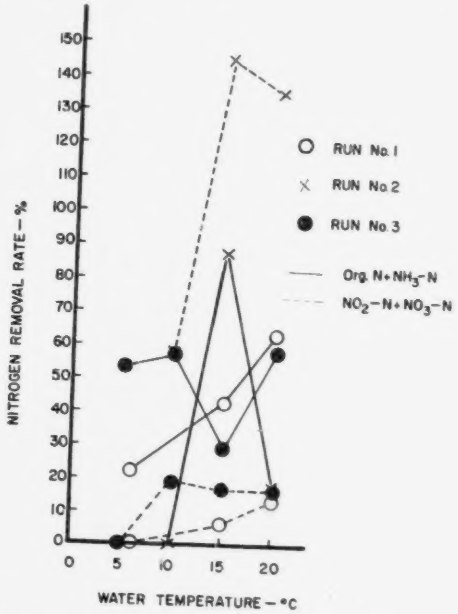


FIGURE 19 Nitrogen removal rate and water temperature

CONCLUSIONS

The following conclusions have been reached from these studies:

1. At low temperatures, the oxygen transfer coefficient in the aeration tank decreases, but the mass of oxygen transfer increases slightly with the lowering of temperature.

2. Sludge volume index increases at low temperatures. It creates a high COD effluent.

3. The value of SVI changes with time after substrate contact with activated sludge. At a low sewage temperature, it is necessary to take sufficiently prolonged aeration time to stabilize the activated sludge which absorbed substrate.

4. The removal ratio of soluble COD is not affected by a decrease of sewage temperature, but total COD of effluent increases due to an increase of suspended matter.

5. The rate of removal of soluble COD varies with water temperature. This relationship is shown in Formula 6, and the temperature coefficient is 1.089.

6. Nitrogen removal is influenced remarkably by temperature. The rate of removal of organic and ammonium nitrogen and the rate of nitrification are shown by Formula 5 and Formula 7 respectively. These rates decrease with lower temperatures, and temperature coefficients are larger than that of removal of COD. This is the reason the effluent of low sewage temperature contains high nitrogen.

ACKNOWLEDGMENT

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MICROBIOLOGIC INDICATORS OF THE EFFICIENCY OF AN AERATED, CONTINUOUS-DISCHARGE, SEWAGE LAGOON IN NORTHERN CLIMATES

John W. Vennes and Otmar O. Olson

INTRODUCTION

Sewage stabilization lagoons have been used in the United States as a method of waste treatment for the past 20 years. The first designed sewage stabilization lagoon in North Dakota was approved by the State Department of Health in 1948 (Van Heuvelen, et al., 1960). The design criteria established for this system were much the same as present standards of 20 lbs. BOD_5/acre/day with a minimum of 100 days retention for winter storage.

The BOD_5 loadings that can be adequately stabilized under aerobic conditions, with oxygen by algal photosynthesis being the primary contributor to aerobiasis, depend on solar radiation, temperature, sewage characteristics, wind action and depth. Temperature and solar radiation are of prime consideration as regards the growth of algae and bacteria. Mixing and distribution of waste in a conventional lagoon are primarily dependent upon wind action. A common problem in conventional lagoons occurs during the first warm days in the spring when anaerobic conditions prevail due to benthic deposits rising to the upper layers of liquid. The recognition of problems encountered in maintaining aerobic conditions in these conventional lagoons led to the application of artificial aeration.

The aerated lagoon is one of the most recent developments in the biological waste treatment system. This concept was initially developed to supplement oxygen during the period of spring break-up by artificially aerating stabilization ponds. Lagoon aeration technology has been developed to the point where aerated lagoons are now designed for complete waste treatment (Sawyer, 1968).

METHODS

The city of Harvey, North Dakota, treats its domestic waste in an aerated lagoon system. The system consists of two cells, each having a water surface area of 1½ acres. The cells are designed to operate in series, each providing a 20-day retention time. The dikes have a 3 to 1 slope and support a 10-foot berm, which allows the complete system to be constructed in less than 6 acres. Overflow manholes in the two cells allow them to be operated at a 10-foot depth with a continuous-discharge to the Sheyenne River.

The aeration system consists of two 15 hp Sutterbilt blowers, each capable of providing 270 cfm at 9 psi. The air is distributed to the two cells through a plastic header pipe paralleling the cells with weighted diffuser tubes connected to the header. There are 78 diffuser lines distributing air into the primary cell and 40 lines in the second cell, giving a total of over 15,000 feet of tubing disbursing the air. The equipment was supplied by Hinde Engineering Company, Highland Park, Illinois, and is known as an Air-Aqua System.

The blowers are operated alternately with a time clock to assure continuous operation of the waste treatment facility. The blowers have been in essentially continuous operation since October 1965.

Ice cover usually is seen in the secondary cell in November or early December while the primary cell does not show ice cover until late December or early January.

The primary cell was designed to be loaded at approximately 400 lbs. of 5-day biochemical oxygen demand (BOD_5)/acre/day and the secondary cell at about 100 lbs. of BOD_5/acre/day. Each cell has a detention time of about 20 days.

Sampling Procedure

Sampling of 24-hour composite raw city waste and effluent from the primary and secondary cell manholes was carried out during the period from January 1966 through June 1969. Daily flows of sewage averaged 175,000 gpd with changes occurring during spring run-off and rainy weather. Samples were usually collected on a monthly basis, however, on occasion two samples were taken during the same month. Since this facility is about 150 miles from the laboratory, the samples were maintained at approximately lagoon temperature in an insulated container and transmitted to the laboratory the same day. Microbiologic determinations were carried out immediately on arrival at the laboratory while the determinations for other parameters were made either the same day or the following morning. Samples in the laboratory were maintained at 5° C prior to the following day determinations.

Laboratory Determinations

Determinations for coliform, fecal coliform, and enterococci were made by the Millipore filter technique while the total bacteria populations were enumerated with tryptone-glucose extract agar (**Standard Methods**, 1965). The determinations for BOD_5, total and suspended solids, pH and Kjeldahl nitrogen were made according to **Standard Methods** (1965). A few determinations were made for biotin by the bacteriologic assay method (Wright and Skeggs, 1944) with *Lactobacillus plantarum*. Several determinations for sulfide and phosphate were also made by using **Standard Methods** (1965).

RESULTS

Laboratory determinations for BOD_5, Kjeldahl nitrogen, pH, total and suspended solids, coliform, fecal coliform, enterococci, and total bacteria are tabulated in the Appendix (Tables 5-12). Graphic presentation of this data is seen in Figures 1 through 6.

This aerated lagoon discharged its effluent continuously into a water course; therefore it was considered useful to determine the mean values for several parameters monitored during the period of the study. It can be seen in Table 1 that the mean BOD_5 for the period was 53 mg/liter (S.D. ±45 mg/liter). The suspended solids show a mean value of 67 mg/liter (S.D. ±39 mg/liter). Mean nitrogen shows values of 7.7 mg/liter (S.D. ±3 mg/liter). There was considerable variation in the composite raw samples in that the mean BOD_5 of the 24-hour composite raw was 332 mg/liter (S.D. ±422 mg/liter). It is thus apparent that the loading on the primary cell changed rather drastically during the period of the study due primarily to a milk plant.

TABLE 1

Statistical Correlation of Several Lagoon Parameters

			mg/liter	
Parameter		Raw	Primary	Secondary
BOD_5	Mean	332 (263)*	126	53
	S.D.	422 (134)	76	45
TVS	Mean	567 (496)	401	353
	S.D.	424 (118)	94	98
VSS	Mean	191 (166)	118	67
	S.D.	168 (118)	74	39
N_2	Mean	20 (15.5)	11.2	7.7
	S.D.	27 (5.3)	3.8	3.0

*Value obtained by disregarding one apparently non-representative sample.

An attempt was made to assess the effect of changing loadings and temperature on this lagoon by comparing BOD_5 values with coliform, fecal coliform, enterococci, and total bacteria. Figures 7 through 9 display this relationship on a log-log basis and suggest that microbial populations in the secondary lagoon are directly related to the BOD_5 or total loading.

Correlations of the lagoon parameters tabulated during the different periods of the study, namely, for the complete sampling period, for the months when the lagoon liquid temperature was near 0° C (i.e. January through March), and the period when the lagoon was near 10° - 20° C (i.e. June through September) are displayed in Table 2. It can be noted in this table that there is a reasonably good correlation between the BOD_5 and the Kjeldahl nitrogen during all periods of the test. One may also observe that the BOD_5 of the secondary lagoon, during times when the lagoon temperature was near 0° C, showed excellent correlation with coliform and fecal coliform bacteria. Less correlation was seen with the enterococci and BOD_5, and little correlation was noted between BOD_5 and total bacteria. During the summer months, when the lagoon temperatures were between 10° and 20° C, the relations between BOD_5 and coliform, fecal coliform, and enterococci were less attractive. There was, however, during this time period, a reasonably good correlation between the secondary lagoon BOD_5 and the total bacterial population.

The relations between nitrogen and coliform, fecal coliform, and enterococci during the winter months of January through March showed excellent correlation; however, the total bacterial population does not show a direct relation to the total nitrogen in the secondary lagoon. Again, as with the BOD_5 and microbial numbers at 10° - 20° C, there is little correlation between the total nitrogen and enteric bacteria at summer temperatures. There is some correlation between total nitrogen and total bacteria during the summer months.

One of the apparent difficulties with the aeration system, as utilized in this continuous-discharge lagoon, is the fact that precipitation of insoluble salts tends to decrease the porosity in the diffuser tubes and pressures tend to build in the system. Occasional cleaning of the system with hydrochloric acid to solubilize the salts usually relieves the pressure and increases the pore size. It was noted during the treatment year from January 1966 through January 1967 that excellent reductions in BOD_5 were obtained in the lagoon. However, during the treatment year 1967

TABLE 2

Correlation Coefficient of Several Lagoon Parameters

Parameters Compared		Correlation Coefficient Months compared	
	All	Jan. - Mar.	June - Sept.
BOD - nitrogen	0.89	0.89	0.82
BOD - coliform	0.33	0.93	0.10
BOD - fecal coliform	0.13	0.93	0.31
BOD - enterococci	0.73	0.70	0.19
BOD - total bacteria	0.52	0.35	0.79
Nitrogen - coliform	0.48	0.91	0.20
Nitrogen - fecal coliform	0.12	0.96	0.33
Nitrogen - enterococci	0.41	0.90	0.53
Nitrogen - total bacteria	0.43	0.17	0.29

through 1968 a cyclic effect in the secondary lagoon parameters was noted and decreased efficiency (due to apparent lack of delivered oxygen) resulted in rather large amounts of BOD_5 and microbial numbers being discharged to the water course. For example, during the months of January through March in 1968 there was approximately a two-fold increase in the nitrogen and suspended solids in the secondary lagoon. There was nearly a four-fold increase in the BOD_5. There was also an increase of from 1 to 2 logs in the coliform, fecal coliform, and enterococci in the secondary effluent. No consistant increase in the total bacterial population was detected.

DISCUSSION

Two of the most important physical factors in this continuous-discharge lagoon affecting the stabilization of organic molecules, and hence microbial growth, appear to be temperature and delivered oxygen. Since two of the other parameters related to microbial growth are substrate availability and pH, and since it is assumed that raw sewage will usually support adequate growth of microorganisms for near complete stabilization of the organic species present, and since pH varied little from neutrality, the major concern with lagoon efficiency must remain with temperature and oxygen.

This study allows the determination of these effects of temperature and oxygen on biologic stabilization and reflects these findings in the relative abundance of several microbial species and BOD_5. For example, it was noted that at near 0° C the total microbial population did not correlate well with BOD_5 reductions, while reductions in coliform, fecal coliform, and enterococci were directly related to the BOD_5 of the supporting medium. Initial populations of enteric organisms were thus determined by the strength of the raw sewage. This relationship does not remain during summer temperatures of 10° - 20° C. It would thus appear that the rates of utilization of lagoon substrates by enteric organisms are different at different temperatures. However, since the total microbial population varied little at different BOD_5 loadings it may be that changes or die-off rates of coliforms may represent primarily the variable of time. Alternately it may be that BOD_5 is not a useful parameter in assessing relations of enteric organisms to organic loading.

In order to verify the assumption that temperature, pH, and substrate availability did not prevent microbial oxidation it appeared useful to compare those periods when the lagoon was known to be at or very near 0° C with those times when it was at 10° - 20° C (when the reduction in organic species, as reflected by the BOD_5, is expected to be at its most active level).

The mean BOD_5 of the raw composite waste during the months of January through March in the years of 1966, 1967, and 1969 was 312 mg/liter (S.D. ±180 mg/liter), while the effluent from the secondary cell had a mean BOD_5 of 50 mg/liter (S.D. ±23 mg/liter). Assuming flows to average 175,000 gpd, then, as seen in Table 3, about 445 lbs BOD_5/day were loaded in the primary cell and about 188 lbs BOD_5/day were loaded in the secondary cell. A reduction of 84% in BOD_5 was thus noted for a period when temperatures were near 0° C.

The mean BOD_5 of the raw composite waste during the months of January through March in 1968 was 665 mg/liter (S.D. ±950 mg/liter), while the effluent from the secondary cell had a mean of 160 mg/liter (S.D. ±19 mg/liter). Thus the loadings on the primary cell were about 970 lbs BOD_5/day, while 329 lbs BOD_5/day were loaded on the secondary cell. An overall reduction of 75% was obtained in the total BOD_5.

TABLE 3

Loadings in Primary and Secondary Aerated Lagoons at Various Times

		Loadings (Pounds BOD_5/day)	
Year/s (months)	**Primary**	**Secondary**	**Effluent**
1966-67-69 (Jan - Mar)	455	188	73
1968 (Jan - Mar)	970	329	233
All (Jan - Sept)	408	136	41
All (All)	485	265	78
		BOD_5 (mg/l)	
	Raw	**Primary**	**Secondary**
1966-67-69 (Jan - Mar)	312 (±180)	129 (±10)	50 (±23)
1968 (Jan - Mar)	665 (±950)	226 (±65)	160 (±19)
All (Jun - Sept)	280 (±117)	93 (±28)	28 (±23)

However, during January through March in 1968, an average 737 lbs BOD_5/day were removed by the aerated cells, while during the other years studied a mean of 382 lbs BOD_5/day were removed. Of course some of the removal of BOD_5 in the primary cell was probably due to sedimentation, and comparing the secondary cell removal rates may be more relevant to this study. In 1968 the secondary cell removed an average of 115 lbs BOD_5/day, while during the other years of the study a mean of 96 lbs of BOD_5/day were removed during the January through March time period. It appeared at this point that temperature did not determine the rate of removal of BOD_5, since about equal amounts were removed during all time sequences in the secondary cell at or near 0° C.

The mean BOD_5 of the raw composite waste during the months of June through September for all years studied was 280 mg/liter (S.D. ±117 mg/liter), while the effluent from the primary cell had a mean of 93 mg/liter BOD_5 (S.D. ±28 mg/liter) and the secondary cell effluent showed a mean BOD_5 of 28 mg/liter (S.D. ±23 mg/liter). Therefore during this time period the primary cell was loaded at 408 lbs BOD_5/day and the secondary cell at 136 lbs BOD_5/day with 41 lbs BOD_5/day being discharged to the water course. Overall reductions in BOD_5 were about 90%, with 67% occurring in the primary cell. It is interesting to note that the secondary cell of this aerated lagoon removes between 95 and 115 lbs BOD_5 at all temperatures. Thus, as suggested above, temperature was not the limiting parameter in total BOD_5 reduction.

The aeration system used in this continuous-discharge lagoon utilizing a 15 hp blower at 270 cfm at 9 psi delivers oxygen sufficient to satisfy approximately 375 lbs BOD/day (367 to 382 lbs BOD/day range) at all temperatures. Thus the system is capable of removing approximately 1 lb of BOD_5/HP hr. Loss of efficiency in the system by decreased porosity in the plastic tubing is probably not as important as the lack of diffusibility inherent in the design of the system. Additional oxygenation transfer will be required to determine the ultimate efficiency of this system at higher temperatures. It may be that at 0° C the maximum efficiency has already been established by the data presented in this report. If this proves to be a fact, then a basis for aeration capabilities of these lagoons has been further validated.

Allowing the assumption that the maximum efficiency for this system has been established, then coliform, fecal coliform, and enterococci reflect reasonably well the biological degradation of organics at 0° C as contrasted to summer temperatures (10° - 20° C) where little or no correlation exists. Additional parameters will need to be investigated in order to determine those reactions that better reflect the biologic reactions that occur at all temperatures. Since only 1% or less of the total microbial population present in the lagoon is represented by the enteric organisms studied here, it is apparent that other organisms must be studied to better define the role of microbiologic indicators in the efficiency of this sewage treatment system.

One system is being studied in our laboratory which relates to several organisms that thrive in sewage oxidation lagoons. It is concerned with the production and utilization of the B vitamin, biotin.

We have found (Fillipi and Vennes, 1970) that biotin is produced by a number of enteric organisms, *Aerobacter aerogenes* being one of the most productive. Utilization of the vitamin seems to be limited to photosynthetic organisms (although some bacteria have been reported to utilize or at least degrade biotin) and in our study *Chlorella* and probably other algae and several species of purple sulfur bacteria seem to be the most active.

Although production of the vitamin has not been established to be directly related to organic loading of the lagoons, utilization seems to be associated with the production or growth of algae. Data are not sufficient at this time to present a definitive outline of these relations, although Table 4 shows the tabulations in lagoons in eastern North Dakota. It is apparent that no good correlation exists between BOD_5 and biotin in these lagoons. Rather it is suggested, based on our previous findings (Fillipi and Vennes, 1970), that the precursors for biotin production are present in lagoons in different amounts and growth of biotin-producing organisms predominate on this and, we expect, other bases. Utilization however seems to be limited primarily to the presence of algae and thus disappearance of biotin may prove to be directly related to algal numbers.

TABLE 4

Biotin and BOD_5 Relations in Several Oxidation Lagoons

Date	Source	BOD_5 (mg/l)	Biotin (ng/l)
9 Apr 69	Harvey Primary	27	10
	Harvey Secondary	33	20
5 Jun 69	Harvey Primary	58	22
	Harvey Secondary	56	134
21 Apr 69	Grand Forks Secondary - E	119	326
	Grand Forks Secondary - W	82	120
21 May 70	Grand Forks Secondary - E	166	1720
	Grand Forks Secondary - W	82	2070
5 Jun 70	Grand Forks Secondary - E	91	2200
	Grand Forks Secondary - W	39	1950
5 Jun 69	Lakota Secondary	12	18
10 Jul 69	West Fargo Secondary	43	53
26 May 70	East Grand Forks Secondary	35	24
20 May 70	Crookston Secondary - W	310	2275
3 Jun 70	Crookston Secondary - W	166	2950
	Crookston Secondary - E	331	2840
20 May 70	Hillsboro Secondary	20	72
3 Jun 70	Hillsboro Secondary	37	284
20 May 70	Northwood Secondary	27	22
3 Jun 70	Northwood Secondary	30	83
14 May 69	Grafton Secondary	513	936
11 Jun 69	Grafton Secondary	460	3530
25 Jun 69	Grafton Secondary	180	8500
10 Jul 69	Grafton Secondary	169	670
7 Aug 69	Grafton Secondary	156	32

It is suggested then that although BOD_5 and total bacterial numbers are somewhat indicative of utilization of organic substrates in lagoons, more sophisticated studies are needed to interrelate microbial numbers and organic constituents responsible for stabilization of wastes. Biotin production and utilization may be an example of the latter relationship.

SUMMARY

1. The limiting parameter in BOD_5 reduction in an aerated, continuous-discharge lagoon in North Dakota appears to be delivered or utilizable oxygen. No difference in BOD_5 reduction was noted in the secondary lagoon at temperatures varying from 0° to 20° C.

2. Coliform, fecal coliform and enterococcal numbers in the secondary lagoon during winter temperatures of near 0° C were directly related to BOD_5 and total nitrogen. During summer temperatures little correlation between these enteric organisms and BOD_5 and total nitrogen was noted. However, there was a correlation between the total microbial population and BOD_5 at summer temperatures.

3. Although the physical aspects of the aeration system can apparently be improved by additional engineering, the understanding of the biologic system will require extensive study of specific ecologic relationships in the lagoon (biotin production and utilization were used to illustrate the complexity of this system).

ACKNOWLEDGMENTS

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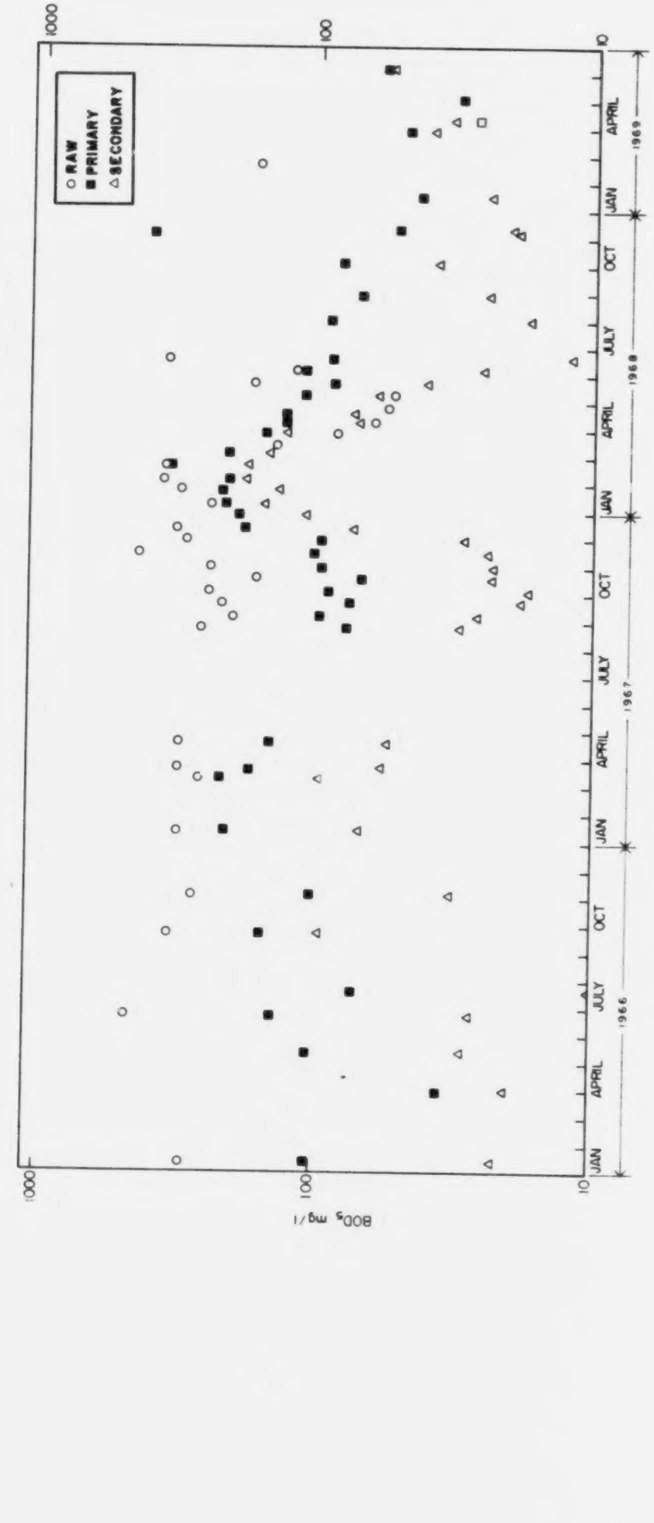

FIGURE 1 BOD$_5$ changes in raw-composite, primary and secondary aerated lagoons

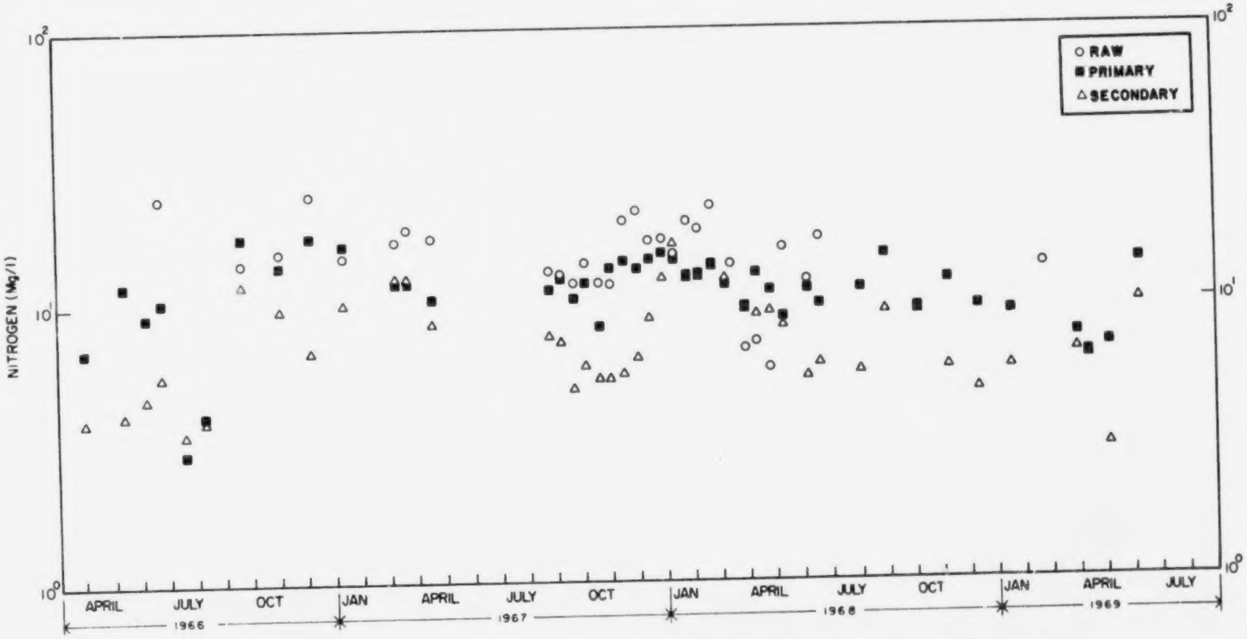

FIGURE 2 Total nitrogen changes in raw-composite, primary and secondary aerated lagoons

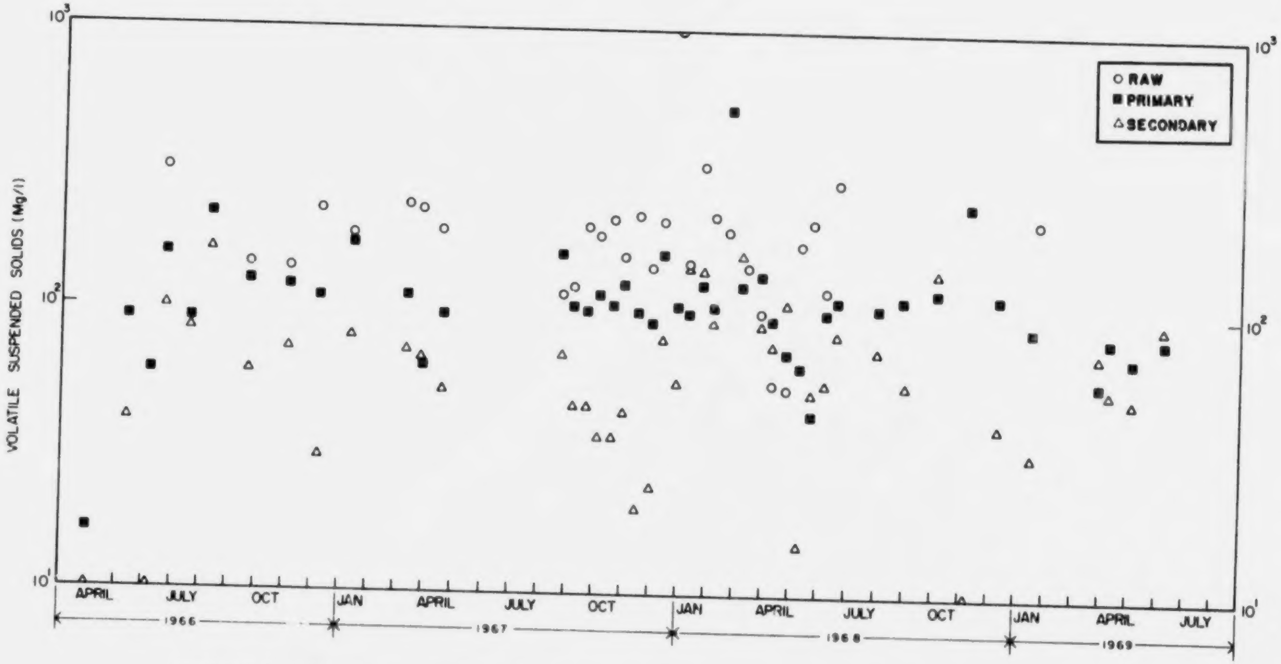

FIGURE 3 Volatile suspended solids changes in raw-composite, primary and secondary aerated lagoons

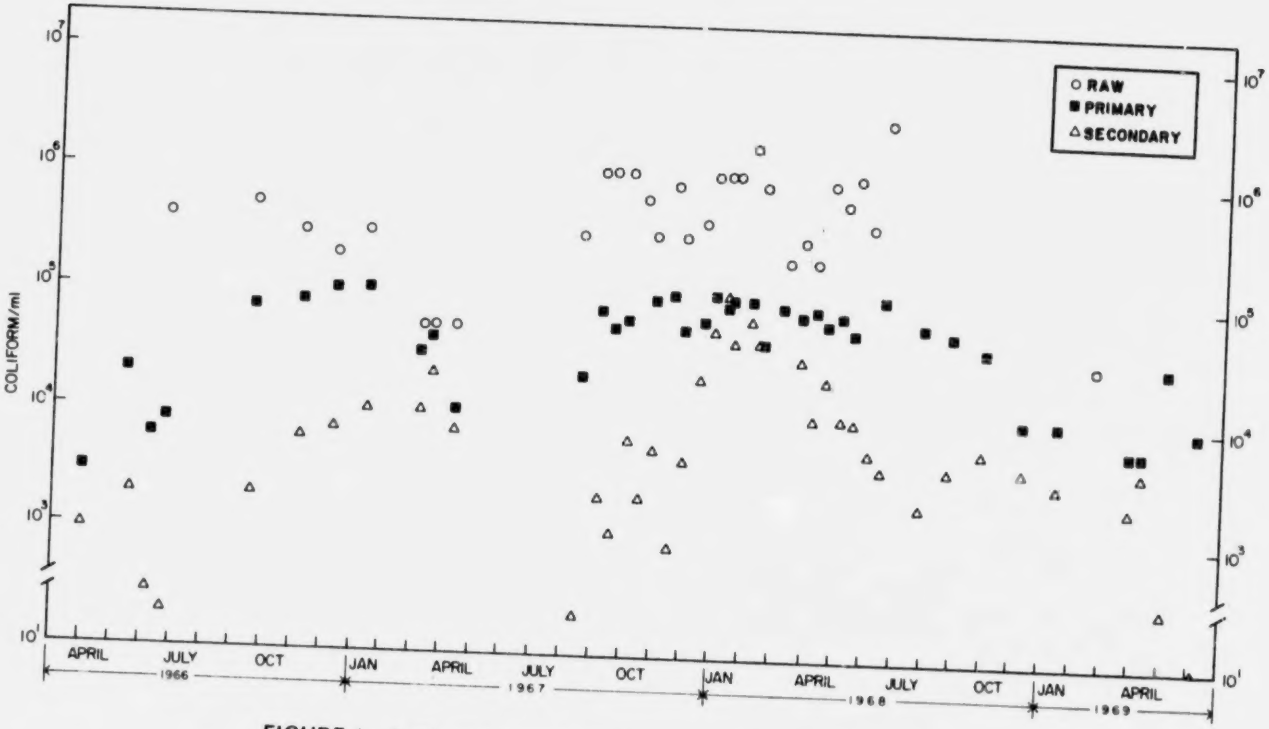

FIGURE 4 Coliform changes in raw-composite, primary and secondary aerated lagoons

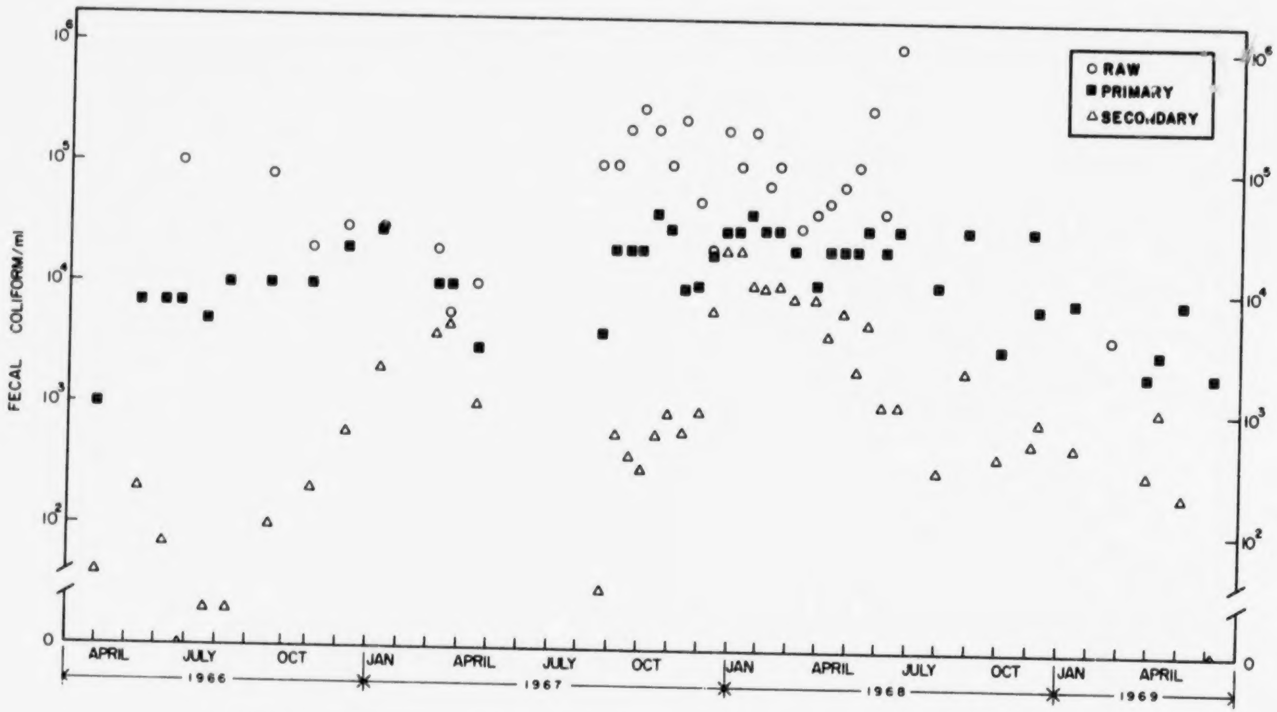

FIGURE 5 Fecal coliform changes in raw-composite, primary and secondary aerated lagoons

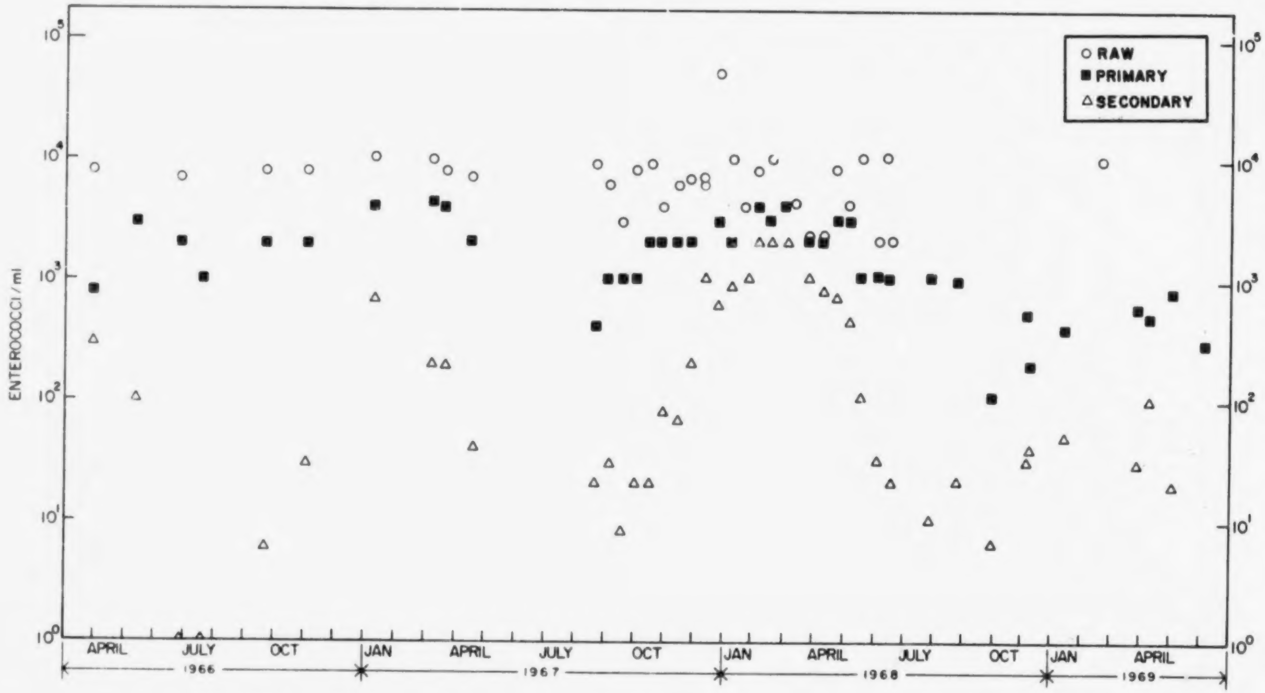

FIGURE 6 Enterococci changes in raw-composite, primary and secondary aerated lagoons

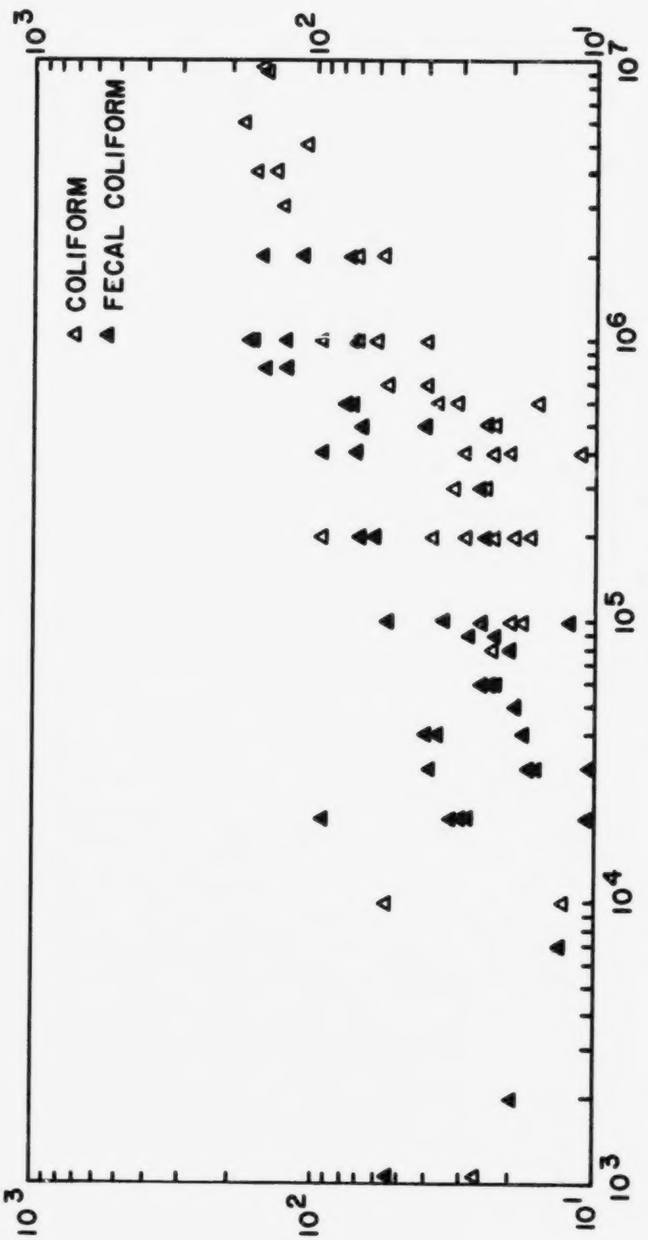

FIGURE 7 Relationship of BOD_5 of secondary lagoon to coliform and fecal coliform organisms

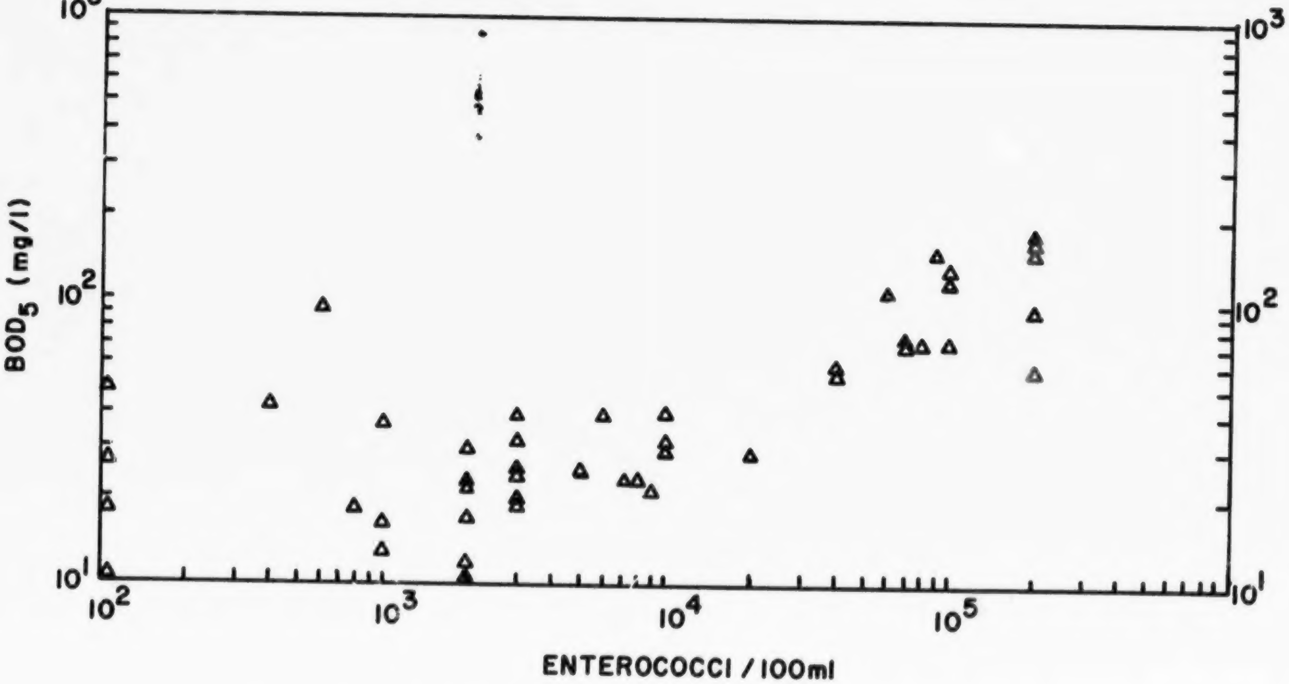

FIGURE 8 Relationship of BOD$_5$ of secondary lagoon to enterococci organisms

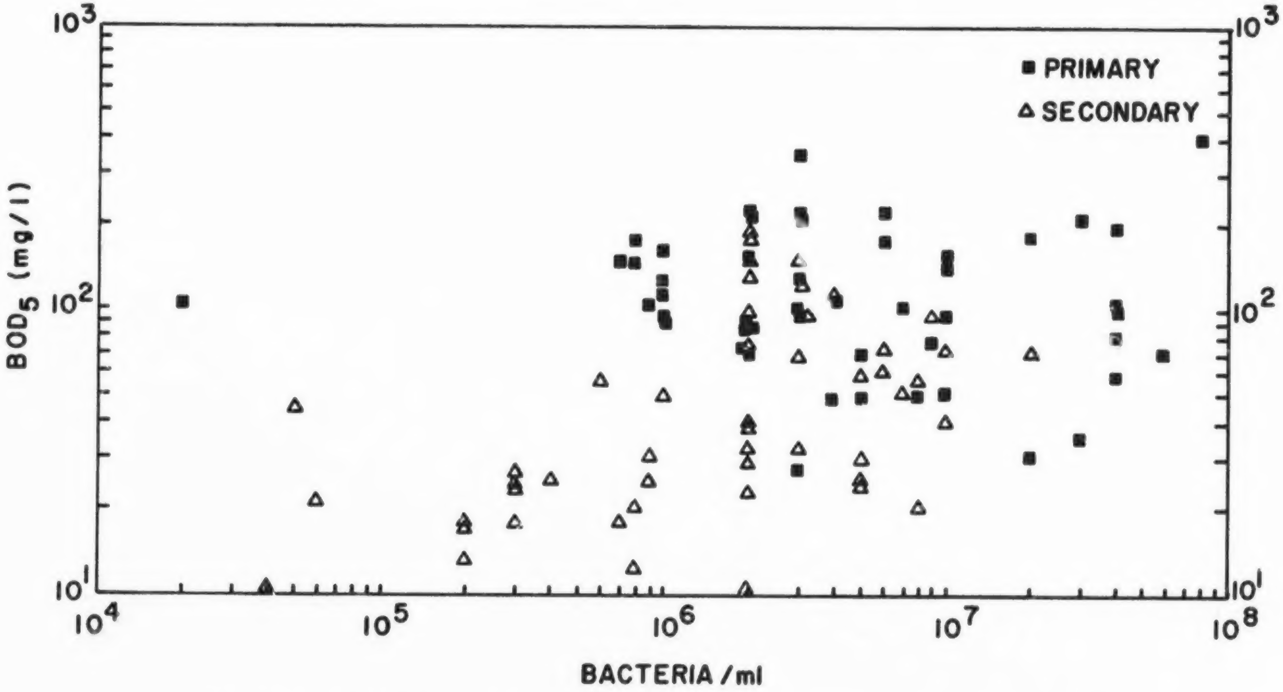

FIGURE 9 Relationship of BOD_5 of primary and secondary lagoon to total bacterial population

APPENDIX

TABLE 5

BOD_5 Changes in Aerated Lagoons

Date	BOD_5 mg/liter		
	Raw	Primary	Secondary
15 Nov 65			25
14 Dec 65		154	16
11 Jan 66	294	104	22
1 Feb 66	692	167	44
8 Feb 66	150	146	48
15 Mar 66		49	52
29 Mar 66		35	20
11 May 66		104	29
6 Jun 66		102	13
22 Jun 66	476	141	27
18 Jul 66		72	8
10 Aug 66		75	18
21 Sep 66	335	155	95
2 Nov 66	275	103	32
7 Dec 66	436	177	40
11 Jan 67	309	209	69
9 Mar 67	262	219	96
22 Mar 67	308	172	57
19 Apr 67	309	146	55
23 Aug 67	263	77	30
6 Sep 67	198	97	26
20 Sep 67	221	76	18
3 Oct 67	245	90	17
18 Oct 67	165	69	23
1 Nov 67	240	96	23
15 Nov 67	440	102	24
29 Nov 67	295	96	29
13 Dec 67	320	181	73
27 Dec 67	2600	193	109
10 Jan 68	240	215	154
24 Jan 68	310	222	137
7 Feb 68	360	207	180
20 Feb 68	345	350	178
5 Mar 68		210	150
12 Mar 68	140		
27 Mar 68	84	154	128
10 Apr 68	62	130	71
24 Apr 68	56	129	74
8 May 68	53	110	60
20 May 68	171	87	40
5 Jun 68	119	112	25
19 Jun 68	348	89	12
30 Jul 68		90	17
27 Aug 68		70	24
2 Oct 68		82	37
6 Nov 68		400	19
9 Dec 68		52	20
14 Jan 69		47	24
21 Feb 69	167		
27 Mar 69		48	39
9 Apr 69		27	33
2 May 69		31	8
5 Jun 69		58	56

TABLE 6

Total Nitrogen Changes in Aerated Lagoons

Date	Raw	Nitrogen mg/liter Primary	Secondary
14 Dec 65		14.7	5.0
11 Jan 66	17.0	12.6	6.9
1 Feb 66	29.2	13.7	7.4
8 Feb 66	12.8	10.9	6.9
15 Mar 66		4.6	5.3
29 Mar 66		6.9	3.8
11 May 66		11.8	4.0
6 Jun 66		9.1	4.6
22 Jun 66	24.3	10.3	5.5
18 Jul 66		2.9	3.4
10 Aug 66		4.0	3.8
21 Sep 66	14.1	17.5	11.8
2 Nov 66	15.4	13.7	9.5
7 Dec 66	24.7	17.5	6.7
11 Jan 67	14.8	16.3	9.9
9 Mar 67	16.7	12.0	12.2
22 Mar 67	18.6	12.0	12.2
19 Apr 67	17.3	10.3	8.4
23 Aug 67	13.1	11.2	7.6
6 Sep 67	12.7	12.7	7.2
20 Sep 67	11.8	10.3	4.9
3 Oct 67	13.9	11.8	5.9
18 Oct 67	11.8	8.7	5.3
1 Nov 67	11.6	13.3	5.3
15 Nov 67	19.8	14.1	5.5
29 Nov 67	20.7	13.1	6.3
13 Dec 67	16.7	14.3	8.7
27 Dec 67	169.0	15.0	12.2
10 Jan 68	15.0	14.3	16.2
24 Jan 68	19.6	12.5	12.5
7 Feb 68	18.4	12.7	12.4
20 Feb 68	22.2	13.6	13.6
5 Mar 68		11.6	12.2
12 Mar 68	13.8		
27 Mar 68	6.8	9.6	9.6
10 Apr 68	7.2	12.8	9.0
24 Apr 68	5.8	11.0	9.2
8 May 68	6.6	8.8	7.8
20 May 68	15.8	8.8	8.2
5 Jun 68	12.0	10.2	5.4
19 Jun 68	17.2	9.8	6.0
30 Jul 68		11.2	5.6
27 Aug 68		14.8	9.2
2 Oct 68		9.4	9.2
6 Nov 68		12.0	5.8
9 Dec 68		9.6	4.8
14 Jan 69		9.2	5.8
21 Feb 69	13.6		
27 Mar 69		7.6	6.6
9 Apr 69		6.4	6.4
2 May 69		7.0	3.0
5 Jun 69		14.2	10.0

TABLE 7

pH Changes in Aerated Lagoons

		pH	
Date	**Raw**	**Primary**	**Secondary**
15 Nov 65			7.9
14 Dec 65		7.7	7.7
11 Jan 66	8.4	7.6	7.7
1 Feb 66	7.5	7.5	7.7
8 Feb 66	8.3	7.7	7.6
15 Mar 66		7.4	7.7
29 Mar 66		7.4	7.4
11 May 66		7.8	8.0
6 Jun 66		7.9	8.1
22 Jun 66	7.4	7.9	8.1
18 Jul 66		7.8	7.8
10 Aug 66		7.7	8.0
21 Sep 66	7.8	8.0	8.1
2 Nov 66	8.1	7.7	7.9
7 Dec 66	8.5	7.6	7.6
11 Jan 67	8.2	7.7	7.7
9 Mar 67	8.5	7.5	7.5
22 Mar 67	8.2	7.5	7.4
19 Apr 67	8.3	7.5	7.6
23 Aug 67	7.9	8.2	8.4
6 Sep 67	8.0	8.1	8.2
20 Sep 67	8.1	8.0	8.1
3 Oct 67	7.5	7.7	7.9
18 Oct 67	8.0	7.8	7.9
1 Nov 67	7.5	7.7	8.0
15 Nov 67	7.4	7.8	7.8
29 Nov 67	7.7	7.7	7.7
13 Dec 67	8.2	7.7	7.6
27 Dec 67	6.9	7.6	7.8
10 Jan 68	7.5	7.4	7.6
24 Jan 68	7.9	7.5	7.6
7 Feb 68	7.6	7.6	7.6
20 Feb 68	7.7	7.2	7.5
5 Mar 68		7.5	7.6
12 Mar 68	8.4		
27 Mar 68	7.6	7.6	7.6
10 Apr 68	7.7	7.7	7.7
24 Apr 68	7.9	7.5	7.7
8 May 68	7.6	7.8	7.9
20 May 68	8.2	7.8	7.9
5 Jun 68	8.7	8.0	8.0
19 Jun 68	7.4	8.0	8.0
30 Jul 68		7.8	7.9
27 Aug 68		7.6	7.9
2 Oct 68		8.4	8.7
6 Nov 68		7.7	8.0
9 Dec 68		7.6	7.8
14 Jan 69		7.4	7.5
27 Jan 69		7.4	7.5
9 Apr 69		7.3	7.4
2 May 69		7.6	8.0
5 Jun 69		7.5	8.0

TABLE 8

Solids Changes in Aerated Lagoons

	Solids mg/liter					
	Raw		Primary		Secondary	
Date	TVS	VSS	TVS	VSS	TVS	VSS
15 Nov 65					301	24
14 Dec 65			436	144	294	36
11 Jan 66	536	220	274	116	363	96
1 Feb 66	676	316	376	148	346	104
8 Feb 66	386	152	418	132	368	52
15 Mar 66			258	80	261	92
29 Mar 66			250	16	184	0
11 May 66			349	92	293	40
6 Jun 66			330	60	273	0
22 Jun 66	766	312	461	156	352	100
18 Jul 66			582	92	530	84
10 Aug 66			528	216	588	160
21 Sep 66	450	136	396	124	400	60
2 Nov 66	444	140	316	120	320	72
7 Dec 66	688	224	450	112	295	30
11 Jan 67	470	184	392	172	332	80
9 Mar 67	480	236	488	112	312	72
22 Mar 67	512	228	278	64	294	68
19 Apr 67	558	192	276	98	235	52
23 Aug 67	490	114	582	160	498	70
6 Sep 67	388	122	474	104	446	46
20 Sep 67	460	200	462	100	486	46
3 Oct 67	406	184	420	116	348	36
18 Oct 67	476	212	418	104	426	36
1 Nov 67	438	164	332	124	346	48
15 Nov 67	590	220	312	100	278	20
29 Nov 67	496	144	268	92	292	24
13 Dec 67	590	204	468	160	370	80
27 Dec 67	2,900	1,024	382	104	342	56
10 Jan 68	558	148	474	100	534	148
24 Jan 68	654	332	428	128	332	140
7 Feb 68	620	220	572	104	392	92
20 Feb 68	588	196	606	524	430	116
5 Mar 68			406	128	418	160
12 Mar 68	392	144				
27 Mar 68	318	100	364	136	364	90
10 Apr 68	298	56	328	94	242	76
24 Apr 68	312	54	340	72	212	108
8 May 68	350	176	364	64	342	16
20 May 68	450	204	328	44	308	52
5 Jun 68	462	120	474	100	508	56
19 Jun 68	676	288	464	112	292	84
30 Jul 68			422	104	528	74
27 Aug 68			422	112	398	56
2 Oct 68			492	120	410	140
6 Nov 68			474	244	202	8
9 Dec 68			348	112	288	40
14 Jan 68			466	88	430	32
21 Jan 69	388	208				
27 Mar 69			280	58	316	72
9 Apr 69			188	82	180	54
2 May 69			390	70	366	50
5 Jun 69			424	84	350	92

TABLE 9

Coliform Changes in Aerated Lagoons

Date	Coliform/100 ml Raw	 Primary	 Secondary
29 Mar 66		4×10^5	1×10^5
11 May 66		2×10^6	2×10^5
6 Jun 66		6×10^5	1×10^4
22 Jun 66	4×10^7	8×10^5	1×10^3
21 Sep 66	5×10^7	7×10^6	2×10^5
2 Nov 66	3×10^7	8×10^6	6×10^5
7 Dec 66	2×10^7	1×10^7	7×10^5
11 Jan 67	3×10^7	1×10^7	1×10^6
9 Mar 67	5×10^6	3×10^6	1×10^6
22 Mar 67	5×10^6	4×10^6	2×10^6
19 Apr 67	5×10^6	2×10^6	7×10^5
23 Aug 67	3×10^7	2×10^6	2×10^4
6 Sep 67	1×10^8	7×10^6	3×10^5
20 Sep 67	1×10^8	7×10^6	1×10^5
3 Oct 67	1×10^8	5×10^6	6×10^5
18 Oct 67	6×10^7	6×10^6	2×10^5
1 Nov 67	5×10^7	9×10^6	5×10^5
15 Nov 67	8×10^7	1×10^7	8×10^4
29 Nov 67	3×10^7	5×10^6	4×10^5
13 Dec 67	4×10^7	6×10^6	2×10^6
27 Dec 67	1×10^8	1×10^7	5×10^6
10 Jan 68	1×10^8	8×10^6	1×10^7
24 Jan 68	1×10^8	9×10^6	4×10^6
7 Feb 68	2×10^8	9×10^6	6×10^6
20 Feb 68	8×10^7	4×10^6	4×10^6
5 Mar 68		8×10^6	9×10^6
12 Mar 68	2×10^7		
27 Mar 68	3×10^7	7×10^6	3×10^6
10 Apr 68	2×10^7	8×10^6	1×10^6
24 Apr 68	9×10^7	6×10^6	2×10^6
8 May 68	6×10^7	7×10^6	1×10^6
20 May 68	1×10^8	5×10^6	1×10^6
5 Jun 68	4×10^7	9×10^7	5×10^5
19 Jun 68	3×10^8	1×10^7	4×10^5
30 Jul 68		6×10^6	2×10^5
27 Aug 68		5×10^6	4×10^5
2 Oct 68		4×10^6	6×10^5
6 Nov 68		9×10^6	2×10^5
9 Dec 68		1×10^6	4×10^5
14 Jan 69		1×10^6	3×10^5
21 Feb 69	3×10^6		
27 Mar 69		6×10^5	2×10^5
9 Apr 69		6×10^5	4×10^5
2 May 69		3×10^6	3×10^4
5 Jun 69		9×10^5	1×10^4

TABLE 10

Fecal Coliform Changes in Aerated Lagoons

	Fecal Coliform/100 ml		
Date	Raw	Primary	Secondary
29 Mar 66		1×10^5	4×10^3
11 May 66		7×10^5	2×10^4
6 Jun 66		7×10^5	7×10^3
22 Jun 66	1×10^7	7×10^5	5×10^2
18 Jul 66		5×10^5	2×10^3
10 Aug 66		1×10^6	2×10^3
21 Sep 66	8×10^6	1×10^6	2×10^4
2 Nov 66	2×10^6	1×10^6	2×10^4
7 Dec 66	3×10^6	2×10^6	6×10^4
11 Jan 67	3×10^6	3×10^6	2×10^5
9 Mar 67	2×10^6	1×10^6	4×10^5
22 Mar 67	6×10^5	1×10^6	5×10^5
19 Apr 67	1×10^6	3×10^6	1×10^5
23 Aug 67	1×10^7	4×10^5	3×10^2
6 Sep 67	1×10^7	2×10^6	6×10^4
20 Sep 67	2×10^7	2×10^6	4×10^4
3 Oct 67	3×10^7	2×10^6	3×10^4
18 Oct 67	2×10^7	4×10^6	6×10^4
1 Nov 67	1×10^7	3×10^6	9×10^4
15 Nov 67	3×10^7	9×10^5	6×10^4
29 Nov 67	5×10^6	1×10^6	9×10^4
13 Dec 67	2×10^6	2×10^6	6×10^5
27 Dec 67	2×10^7	3×10^6	2×10^6
10 Jan 68	1×10^7	3×10^6	2×10^6
24 Jan 68	2×10^7	4×10^6	1×10^6
7 Feb 68	7×10^6	3×10^6	1×10^6
20 Feb 68	1×10^7	3×10^6	1×10^6
5 Mar 68		2×10^6	8×10^5
12 Mar 68	3×10^6		
27 Mar 68	4×10^6	1×10^6	8×10^5
10 Apr 68	5×10^6	2×10^6	4×10^5
24 Apr 68	7×10^6	2×10^6	6×10^5
8 May 68	1×10^7	2×10^6	2×10^5
20 May 68	3×10^7	3×10^6	5×10^5
5 Jun 68	4×10^6	2×10^6	1×10^5
19 Jun 68	1×10^8	3×10^6	1×10^5
30 Jul 68		1×10^6	3×10^4
27 Aug 68		3×10^6	2×10^5
2 Oct 68		3×10^5	4×10^4
6 Nov 68		3×10^6	5×10^4
9 Dec 68		7×10^5	8×10^4
14 Jan 69		8×10^5	5×10^4
21 Feb 69	4×10^5		
27 Mar 69		2×10^5	3×10^4
9 Apr 69		3×10^5	1×10^5
2 May 69		8×10^5	2×10^4
5 Jun 69		2×10^5	1×10^3

TABLE 11

Enterococci Changes in Aerated Lagoons

	Enterococci/100 ml		
Date	Raw	Primary	Secondary
11 Jan 66	4×10^4	1×10^4	9×10^3
1 Feb 66	2×10^4	4×10^2	4×10^2
8 Feb 66	1×10^2	5×10^2	1×10^2
15 Mar 66	7×10^4	1×10^5	
29 Mar 66	8×10^5	8×10^4	3×10^3
11 May 66		3×10^5	1×10^4
6 Jun 66		2×10^5	1×10^3
22 Jun 66	7×10^5	2×10^5	1×10^2
18 Jul 66		1×10^5	1×10^2
10 Aug 66		1×10^5	1×10^2
21 Sep 66	8×10^5	2×10^5	6×10^2
2 Nov 66	8×10^5	2×10^5	3×10^3
7 Dec 66	9×10^5	6×10^5	6×10^3
11 Jan 67	2×10^6	4×10^5	7×10^4
9 Mar 67	1×10^6	5×10^5	2×10^5
22 Mar 67	8×10^5	4×10^5	2×10^5
19 Apr 67	7×10^5	2×10^5	4×10^4
23 Aug 67	9×10^5	4×10^4	2×10^3
6 Sep 67	6×10^5	1×10^5	3×10^3
20 Sep 67	3×10^5	1×10^5	8×10^2
3 Oct 67	8×10^5	1×10^5	2×10^3
18 Oct 67	9×10^5	2×10^5	2×10^3
1 Nov 67	4×10^5	2×10^5	8×10^3
15 Nov 67	6×10^5	2×10^5	7×10^3
29 Nov 67	7×10^5	2×10^5	2×10^4
13 Dec 67	6×10^5	7×10^5	1×10^5
27 Dec 67	5×10^6	3×10^5	6×10^4
10 Jan 68	1×10^6	2×10^5	9×10^4
24 Jan 68	4×10^5	4×10^5	1×10^5
7 Feb 68	8×10^5	4×10^5	2×10^5
20 Feb 68	1×10^6	3×10^5	2×10^5
5 Mar 68		4×10^5	2×10^5
12 Mar 68	4×10^5		
27 Mar 68	2×10^5	2×10^5	1×10^5
10 Apr 68	2×10^5	2×10^5	8×10^4
24 Apr 68	8×10^5	3×10^5	7×10^4
8 May 68	4×10^5	3×10^5	4×10^4
20 May 68	1×10^6	1×10^5	1×10^4
5 Jun 68	2×10^5	1×10^5	3×10^3
19 Jun 68	2×10^5	1×10^5	2×10^3
30 Jul 68		1×10^5	1×10^3
27 Aug 68		9×10^4	2×10^3
2 Oct 68		1×10^4	1×10^3
6 Nov 68		5×10^4	3×10^3
9 Dec 68		2×10^4	4×10^3
14 Jan 69		4×10^4	5×10^3
21 Feb 69	1×10^6		
27 Mar 69		6×10^4	3×10^3
9 Apr 69		5×10^4	1×10^4
2 May 69		8×10^4	2×10^3
5 Jun 69		3×10^4	

TABLE 12

Total Bacterial Changes in Aerated Lagoons

	Bacteria/100 ml		
Date	Raw	Primary	Secondary
11 Jan 66	3×10^6	2×10^6	6×10^6
1 Feb 66	7×10^7	1×10^8	5×10^6
8 Feb 66	4×10^7	8×10^7	1×10^8
15 Mar 66		8×10^8	7×10^8
29 Mar 66	3×10^8	3×10^9	8×10^8
11 May 66		4×10^9	5×10^8
6 Jun 66		9×10^7	2×10^7
22 Jun 66	2×10^9	1×10^9	3×10^7
18 Jul 66		2×10^8	4×10^6
10 Aug 66		2×10^8	2×10^7
21 Sep 66	9×10^8	1×10^9	9×10^8
2 Nov 66	9×10^8	4×10^9	3×10^8
7 Dec 66	5×10^8	6×10^8	2×10^8
11 Jan 67	4×10^8	3×10^8	3×10^8
9 Mar 67	3×10^8	2×10^8	3×10^8
22 Mar 67	3×10^8	8×10^7	5×10^8
19 Apr 67	5×10^8	7×10^7	8×10^8
23 Aug 67	2×10^9	2×10^8	9×10^7
6 Sep 67	4×10^9	3×10^8	4×10^7
20 Sep 67	3×10^9	9×10^8	7×10^7
3 Oct 67	3×10^9	2×10^8	2×10^7
18 Oct 67	2×10^9	5×10^8	3×10^7
1 Nov 67	2×10^9	1×10^9	2×10^8
15 Nov 67	3×10^9	7×10^8	3×10^7
29 Nov 67	2×10^9	3×10^8	2×10^8
13 Dec 67	1×10^9	2×10^9	1×10^9
27 Dec 67	9×10^9	4×10^8	4×10^8
10 Jan 68	2×10^9	3×10^8	3×10^8
24 Jan 68	5×10^9	6×10^8	2×10^8
7 Feb 68	3×10^9	3×10^9	2×10^8
20 Feb 68	2×10^9	3×10^8	2×10^8
5 Mar 68	9×10^8	2×10^8	2×10^8
27 Mar 68	2×10^9	2×10^8	3×10^8
10 Apr 68	2×10^9	3×10^8	2×10^9
24 Apr 68	2×10^9	1×10^8	6×10^8
8 May 68	2×10^9	4×10^8	6×10^8
20 May 68	2×10^9	1×10^8	1×10^9
5 Jun 68	6×10^8	1×10^8	5×10^8
19 Jun 68	4×10^9	2×10^8	8×10^7
30 Jul 68		1×10^8	3×10^7
27 Aug 68		6×10^9	9×10^7
2 Oct 68		4×10^9	4×10^8
6 Nov 68		8×10^9	2×10^8
9 Dec 68		1×10^9	8×10^7
14 Jan 69		5×10^8	9×10^7
21 Feb 69	2×10^8		
27 Mar 69		4×10^8	2×10^8
9 Apr 69		3×10^8	2×10^8
2 May 69		2×10^9	2×10^8
5 Jun 69		4×10^9	6×10^7

DISINFECTION AND TEMPERATURE INFLUENCES

Cecil Chambers and Gerald Berg

INTRODUCTION

The purpose of this discussion is to explore a number of disinfection methods, some conventional, and some not so conventional, and to examine their potential strengths and weaknesses for disinfecting effluents under cold climate conditions. A second objective will be to outline the need for research to assist in developing guidelines for the most efficient practical use of disinfectants where extremely low temperatures prevail.

Disinfection, like primary settling, aeration, and other steps in waste treatment, cannot be considered independently of other phases of the process. All of these factors affect the quality of the final effluent produced. This is especially important in the situation we are concerned with, because the arctic environment influences the various methods of waste treatment in ways which have profound effects on disinfection efficiency.

In general, effluents in arctic areas are relatively crude. A majority of the treatment plants in Alaska have only primary settling. The high turbidity and organic content of such effluents make them difficult to disinfect by any method available. In addition, because biological oxidation proceeds at a slow pace at low temperatures, the effluents from secondary activated sludge plants and other biological treatment processes are incompletely oxidized and high in ammonia nitrogen content.

Because its primary purpose is the prevention of transmission of disease, disinfection is considered the most critical step in waste treatment. Nevertheless, application of massive doses of chlorine, for example, should not be accepted as a substitute for adequate removal of solids and stabilization of soluble organic matter. When such crude effluents are discharged to receiving waters, the low biological activity at arctic temperatures will contribute to an increasing and cumulative load of oxidizable material, especially in lakes, streams, and estuarine waters.

Dissolved oxygen levels frequently reach extremely low concentrations under ice cover in Alaska. Under these circumstances, any attempt to circumvent adequate treatment by excessive application of disinfectants will further deplete the critically low dissolved oxygen supply. In addition, the question is being asked with increasing frequency, are we, in applying massive doses of chlorine to crude effluents, creating toxic chlorine addition products that adversely affect the flora and fauna in the receiving water? What is the effect on the aerobic food chain? When such possible toxicity is superimposed on the effect of an already precariously low oxygen level, it may lead to the demise of a critical segment of the food chain. This can be especially important in cold climate areas where biological productivity of waters is low. The consideration of non-residual disinfectants may have potential to resolve or minimize this problem in some low temperature situations. Another approach is neutralization of the disinfectant before discharge or selection of a chemical disinfectant that is effective at minimum residual levels.

The efficiency of chemical disinfectants decreases with decreasing temperature (Clarke and Chang, 1959). Another problem that should be considered is the extended survival of organisms at near-freezing water temperatures. Both enteropathogenic and coliform bacteria are known to survive in water for extended periods of time at very low temperatures, and viruses survive for long periods when frozen in ice. Clark (1970) has pointed out that in Alaska, "Stream and lake waters will be used for drinking water for some time to come without treatment in spite of all efforts to provide, require, and encourage treatment."

Because of prolonged survival of pathogenic organisms in cold water and ice, the organisms may be transported by currents under the ice and by drifting ice when break-up time comes to serve as sources of infection far from their point of origin. In addition, there is widespread groundwater contamination in many areas especially during spring thaws. This is further aggravated by inadequate disinfection of wastes and extended survival of contaminants at low groundwater temperatures which are frequently just above the freezing point. These conditions combined with the widespread practice of drinking untreated water emphasize the importance of thorough disinfection of effluents in cold climate areas.

GENERAL AND BACTERICIDAL CONSIDERATIONS

Chlorine

Chlorine is more widely used for disinfection of water and wastewater than any other chemical agent. Regardless of whether chlorine is added to water as liquid chlorine (Cl_2), sodium hypochlorite (NaOCl), or calcium hypochlorite ($Ca(OCl)_2$), the same disinfecting entities will be produced (Fair et al., 1948). In wastewater disinfection, two forms of chlorine are of primary interest: they are (1) hypochlorous acid (HOCl) and (2) chloramines.

A third form, hypochlorite ion (OCl^-), has little if any disinfecting efficiency. The hydrolysis product of the reaction of chlorine and water, HOCl, is the predominant form in which chlorine exists in aqueous solution between pH levels of 2.0 and 7.0. Between pH 7.0 and 8.0 progressive ionization of the HOCl takes place. At the higher pH level most of the chlorine exist as OCl^- ions.

Significant amounts of ammonia-nitrogen, probably 20 ppm or more, can be anticipated in arctic effluents. This ammonia will combine with chlorine to produce chloramines of which monochloramine is the dominant, and probably the most germicidal, form of chlorine that will be present at the pH of most effluents (Baker, 1959). The exception would be break-point chlorination, where ammonia-nitrogen is destroyed by application of chlorine at a chlorine-to-nitrogen ratio of approximately 9 to 1 (Butterfield, 1948a) to provide residual HOCl. Chlorine demand, caused by organic components of sewage, is a serious problem in disinfection of wastewater. Assuming no chlorine demand, approximately 180 ppm of titratable chlorine would be required to reach the breakpoint and provide residual HOCl at an ammonia-nitrogen concentration of 20 ppm. Except initially, HOCl is not likely to be present in most chlorinated effluents.

Monochloramine is a relatively inefficient disinfectant when compared to HOCl (Butterfield, 1948b). Butterfield and Wattie (1946) in tests with chloramine* using *Escherichia coli*, *Salmonella*

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*While the term chloramine is used in their report, conditions were such that monochloramine was the dominant form produced.

typhosa, and *Shigella sonnei* as test organisms, concluded that, "A reduction of 20° C in temperature (20° C - 25° C to 2° C - 6° C) requires 9 times the exposure period, or 2.5 times as much chloramine to produce a 100% kill." Decreasing the hydrogen ion concentration decreases the disinfecting efficiency of both HOCl and monochloramine.

When the effects of an unfavorable temperature and pH are combined, the disinfecting efficiency of monochloramine is very seriously depressed; these effects are demonstrated by Butterfield's (1948b) data which are presented in Figure 1. The advantages of maintaining the lowest possible pH, consistent with avoiding corrosion problems, are readily apparent when using HOCl or monochloramine as a disinfectant at low temperature.

Iodine

At acid pH levels, iodine exists primarily in the elemental state and forms tri-odide and higher iodides in the presence of increasing quantities of iodide ions. At increasing pH levels, elemental iodine hydrolyzes to hypoiodous acid (HOI) which, in the absence of excessive iodide ions, may constitute about 40% of the iodine at pH 8.0. As the pH increases beyond this level, the HOI decomposes to iodates which are essentially non-germicidal. Elemental iodine and several of its derivatives are efficient disinfectants, although not quite as efficient as HOCl. While iodine is more expensive than chlorine, it does not form iodamines and has relatively low reactivity to organic matter (Chang, 1966). Accordingly, iodine has considerable potential for circumventing some of the problems encountered with chlorine.

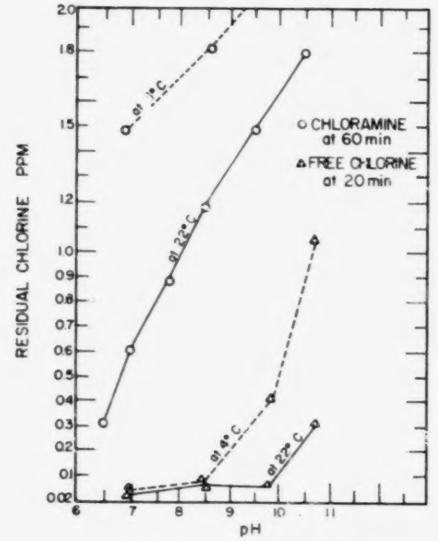

FIGURE 1 The combined effort of variations in pH and temperature on the disinfecting efficiency of free chlorine and chloramine. After Butterfield (1948b)

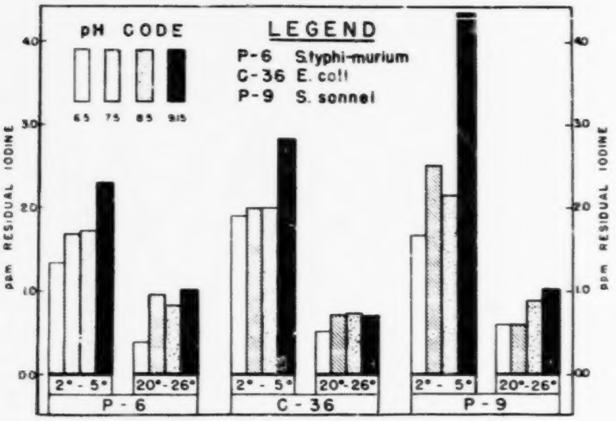

FIGURE 2 Average ppm iodine required to kill all test bacteria in 1 minute. After Chambers et al. (1952)

Chambers et al. (1952) in tests with *E. coli, Aerobacter aerogenes, Streptococcus fecalis,* and three species of *Salmonellae*, including *S. typhosa*, and four species of *Shigellae*, concluded that, "At a given exposure time and pH the iodine concentration required to kill at 2° C to 5° C may be as much as four times greater than at 20° C to 26° C," and ". . . increases in pH reduce the bactericidal action of iodine." These effects are illustrated in Figure 2 which presents the results obtained with a pollution index organism, *E. coli*, and the two most resistant enteropathogenic bacteria tested, *Salmonella typhimurium* and *S. sonnei*. Recent information indicates that the most desirable pH range for disinfection with iodine is pH 7.0 to 8.0 (Berg, 1970).

Because iodine has not been widely used for disinfection of water and wastewater, methods of application are not as universally understood as is the case with chlorine. Chang (1966) has designed a system for application of iodine to water that can be adapted easily to disinfection of effluents. His report also presents an excellent bibliography on disinfection of water with iodine. In addition, a patent has been granted on a system designed to convert iodine to the gaseous state for use in disinfection of water (Starbuck, 1967).*

Excess Lime

In water and wastewater treatment, lime is usually considered primarily as a flocculating agent. While there are some problems such as sludge disposal associated with the use of lime, lime flocculation can markedly reduce the solids content of raw sewage and the overall BOD reduction may be as much as 70% (Bishop and Sanoworth, 1970).

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*References to commercial products are not to be construed as endorsement by the Federal Water Quality Administration.

Flocculation with lime can remove or destroy a relatively high percentage of the microorganisms present. What is not generally realized is that lime can also be used as a disinfectant if pH and contact time are maintained at suitable levels, and it is not subject to the organic demand problems of halogen disinfectants and ozone. The data of Wattie and Chambers (1942), presented in Figure 3 show the disinfecting efficiency of increasingly high pH at temperatures of 0° C to 1° C and 20° C to 25° C with *E. coli*.

Results obtained by the same people in comparable tests with *S. typhosa* are presented in Figure 4. Riehl et al. (1952) augmented this work in subsequent studies with a wider spectrum of pathogenic organisms at several temperatures ranging from 2° C to 25° C. Their results are comparable to those presented in Figures 3 and 4.

Ozone

Although ozone is widely used as a disinfectant throughout Europe, its primary use for treating drinking water in the United States is for the removal of taste, odor, and color.

There is very little information in the literature relating to the possibility of using ozone as a disinfectant for waste treatment plant effluents. Stumm (1958), however, stated ". . . ozone has to be considered as a potential disinfectant for water and sewage." He also indicated the need for research in his concluding statement ". . . more investigations are needed to evaluate the applicability of ozone as a practical tool for the sanitary engineer in water treatment and waste disposal." Recently Smith (1967) presented data obtained in studies with effluent from a municipal activated sludge plant. Tests were run with non-nitrified effluent, nitrified effluent, raw

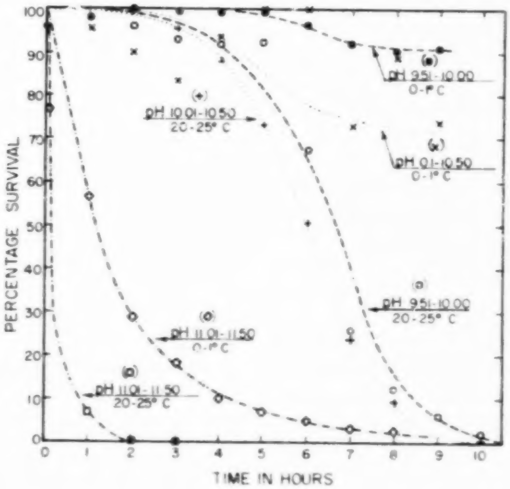

FIGURE 3 **Death rate of *Escherichia coli* at various pH ranges at 0° - 1.0° and 20° - 25° C. After Wattie and Chambers (1942)**

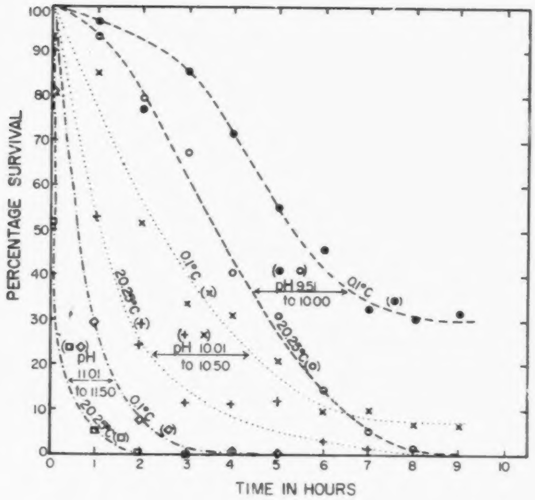

FIGURE 4 Death rate of *Salmonella typhosa* at various pH ranges at 0° - 1° and 20° - 25° C. After Wattie and Chambers (1942)

sewage filtered through glass wool, and raw sewage (aluminum sulfate precipitated). They concluded that disinfection at the 99% kill level was not attained with secondary effluents, and results with raw sewage were less acceptable. In subsequent laboratory pilot scale studies they concluded that ". . . an ozone dose of 7 to 10 ppm may be sufficient for disinfection of municipal sewage treatment plant effluent under these circumstances."

In considering the possible applicability of ozone for disinfection of wastes under cold climate conditions, the fact that ozone is a very powerful oxidizing agent makes it susceptible to organic demand. This characteristic is offset to a degree by the fact that some reduction in BOD occurs and the end products of the oxidation are non-toxic. In addition, ozone is rapidly dissipated and no toxic residual disinfectant remains in the effluent. A distinct advantage is the fact that ozone is not susceptible to reductions in efficiency in the presence of ammonia, as is the case with chlorine. Increased solubility at the low temperatures encountered in the Arctic may also be an advantage. However, the excessively high iron content of some Alaskan groundwaters (6 to 14 mg/l common; Clark, 1970) may be a problem if the iron is in the reduced state and such waters find their way into the disposal system.

Bromine

Bromine is recognized as a powerful bactericidal agent. Although it is somewhat more expensive than chlorine, bromine has been used with considerable success as a swimming pool disinfectant in Illinois (Klassen and Sieg, 1948). Nevertheless, until recently, little attention has been given to investigation of bromine for use in disinfection of water and even less consideration has been given to the potential of bromine for disinfection of wastewater. The work of Johannesson (1960) has provided a basis for a better understanding of the fundamental behavior of bromine as a water disinfectant, and this has served as a stimulus to increased interest in bromine. Because bromamines are essentially as germicidal as free bromine, a highly potent disinfectant, bromine may have potential for use in disinfection of waste treatment plant effluents especially in cold climate areas where effluents with high ammonia content may be expected to be a continuing problem. Bromine has the disadvantage, however, of being relatively susceptible to organic demand and elevated pH (9.0-9.5) is necessary to produce significant amounts of bromamines.

The Federal Water Quality Administration has been sponsoring research to determine the relative efficiency of chlorine and bromine for disinfection of activated sludge plant effluent (Sollo et al., 1970). In this work, both dosage and residuals of chlorine and bromine are being correlated with percent survival of coliforms, fecal coliforms, and total bacterial numbers. Anomalous results have been obtained with chlorine at 30° C (pH 9.0) and these results cannot be explained. Control of bromine concentrations has proven difficult because of the rapid bromine decay rate. Combinations of chlorine and bromine are also being evaluated, and it appears that pretreatment with chlorine, followed by bromine, may produce effective disinfection at a lower cost than bromine alone. The preliminary results obtained indicate that ". . . the efficiency of disinfection is almost independent of the chlorine dosage, so long as it is greater than the immediate demand. Thus, the function of the clorine appears to be only that of conserving bromine by satisfying demand." The minimum test temperatures in this project were 10° C, but this work is being continued and tests at 0° C to 2° C are planned for the future.

Pasteurization

Pasteurization, because it is unaffected by the chemical quality of wastes, may be feasible in arctic climates in some locations. Sludges and effluents from sanitoria have been disinfected with heat in a number of European countries and sewage sludge has been pasteurized before spraying on agricultural land in Germany (Kugel, 1968). Goldstein et al. (1960) tested a system for pasteurization of water that was effective in destroying coliform organisms. They considered the principal advantages to be reliability and simplicity, both important factors in Alaska. On a household scale, costs were estimated to be approximately $1.00 per 1,000 gallons, but they would have been lower if the unit had been operated constantly. At cold climate temperatures, costs would be somewhat higher and revised design criteria would probably be necessary with wastewater to minimize fouling of heat exchangers. Construction plans for this unit have been published (U.S. Public Health Service, 1959). Seiberling and Harper (1955) have also reported on a pasteurizer design for successful high temperature short time exposure of water. Harper stated (personal communication to C. W. Chambers, 1965) that after approximately 10 years of operation ". . . this unit has been extremely satisfactory. . ."

Miscellaneous Disinfectants

Gamma radiation has been advocated by some people for disinfection of wastes. The fact that it is a physical process might indicate potential for low temperature use, but there appears to be little in the work reported to offer encouragement. There have been reports of a marked synergistic action of gamma radiation on the disinfecting action of chlorine, and this could be advantageous under cold climate conditions; however, the work reported has been inadequately controlled. The authors are investigating this phenomenon and results to date show no evidence of any practically significant synergistic effect with monochloramine when conditions are adequately controlled.

Ultraviolet light has some potential for disinfection of high quality effluents. The types of effluents produced in Alaska may be high in iron and color content; and both of these interfere seriously with the disinfecting efficiency of ultraviolet light. Another trait that has to be considered is the fact that low ambient temperatures reduce lamp efficiency. It is possible that ultraviolet light may have some potential for use with effluents that have been highly clarified by chemical coagulation, provided interference from color and residual lime floc are not encountered.

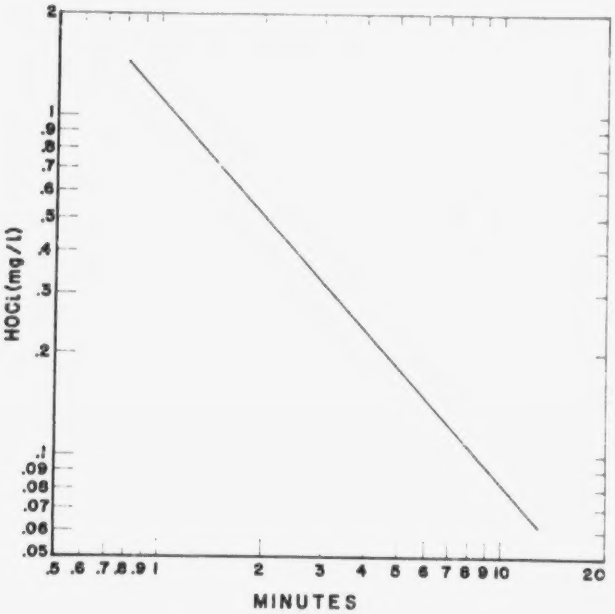

FIGURE 5 Inactivation of poliovirus 1 by HOCl. After Weidenkopf (1958)

VIRUCIDAL CONSIDERATIONS

Chlorine

Hypochlorous acid (HOCl) is a rapid virucide (Fig. 5) (Weidenkopf, 1958), while the OCl^- ion is a poor virucide, if in fact it kills any viruses at all. In a clear aqueous solution, even at low temperature, chlorine is a very rapid virucide at acid pH, and a very slow virucide beyond mildly alkaline pH levels. Chloramines are slow virucides (Fig. 6) (Shuval, 1966), usually too slow for effective treatment of virus-bearing water. Even the quick virucidal activity that occurs immediately after chlorine is added to a sewage effluent, an activity that reflects the action of HOCl before it has all reacted to form chloramines, leaves a large amount of virus to the slow-killing chloramines. Since even very small numbers of viruses are capable of producing infections (Plotkin and Katz, 1967), it is clearly important that all viruses should be removed from any waters with which man may come in contact. Thus, when HOCl cannot be maintained, or when its presence cannot be tolerated, we must consider other disinfectants.

Survival of Viruses in Sewage and Water

Freezing brings about the formation of ice crystals in bacteria, and with this, eventual disruption of the cellular membrane of many cells. The result is a gradual to rapid die-off of bacteria, the rate depending on the species, temperature, and other environmental factors. Viruses are not cellular, and freezing does not destroy them. To the contrary, freezing is the best known method for storing viruses, and the lower the temperature, the better. In the laboratory viruses are preserved by cold-temperature storage. Viruses may be stored at -70° C for a decade or more. Thus, in the cold environment of the North, the normal die-off of viruses experienced in warmer climates probably does not occur.

Figure 7 (Berg, 1966) shows the relationship between temperature and time for the destruction of 99.9% of three different viruses in stored sewage. This graph was prepared from data obtained by others (Clarke, Berg et al., 1962). At 10° C, 99.9% destruction of echovirus 7 took more than 100 days, and both polivirus 1 and echovirus 12 required more than 60 days. Thus, enteroviruses can survive in sewage for many months at low temperatures. In ice, survival time will exceed cold water survivals and, of course, the lower the temperature, the longer will be the survival.

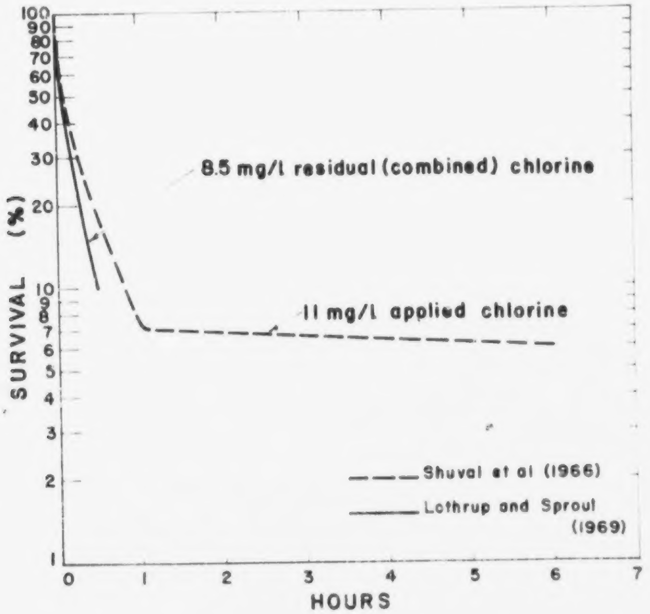

FIGURE 6 Inactivation of poliovirus 1 by applied chlorine

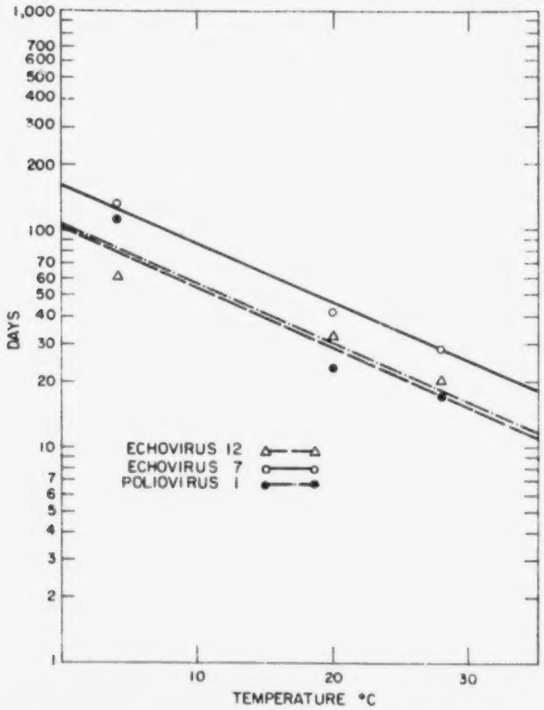

FIGURE 7 Relationship between time and temperature for 99.9% reduction of virus in stored sewage. After Clarke, et al. (1962) as revised by Berg (1966)

Viruses do not survive as long in river water as they do in sewage (Clarke, Berg et al., 1962; Berg and Berman, 1970) and there is some evidence that they do better in lightly polluted water than in heavily polluted water (Clarke, Berg et al., 1962). Their survival in distilled water is also of long duration.

There is no simple explanation for survival patterns of viruses in waters of various qualities. Since autoclaving increases survival in certain river waters, microbial life, their enzymes or other heat-labile matter may account for some of the pattern. Certain divalent cations are known to increase the survival of some viruses, and decrease it for others. Clearly, the chemistry of the water, as well as the temperature, is an important determinant of virus survival.

Iodine

Elemental iodine is a rapid virucide, but not as rapid as HOCl. Elemental iodine, however, can be effective in circumstances that impede the effectiveness of HOCl. In the presence of ammonia, for example, where HOCl reacts to form the relatively slow chloramines, the iodine remains free to react with viruses. Elemental iodine is a faster virucide than chloramine (Berg and Berman, 1970).

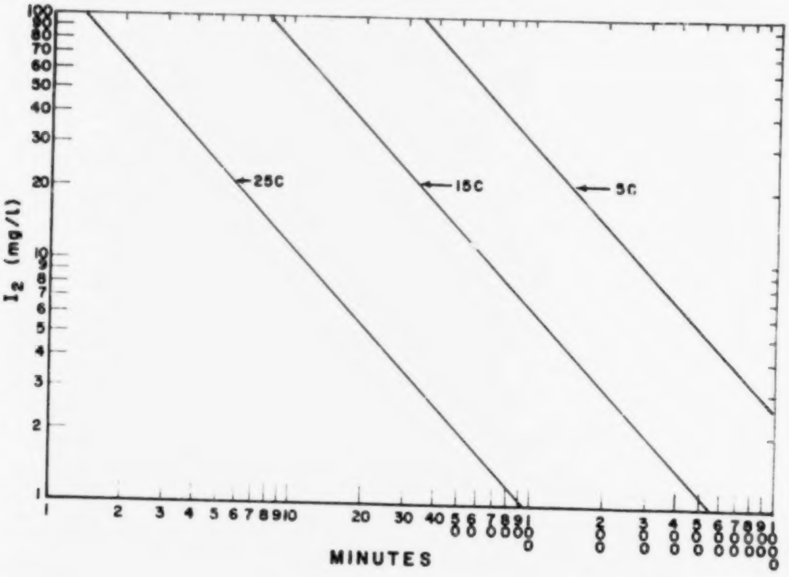

FIGURE 8 Inactivation of coxsakievirus A9 by elemental iodine. After Berg (1964)

At pH 8.0, where most HOCl has ionized to the poorly virucidal OCl^- ion, much of the elemental iodine has hydrolyzed to HOI which is considerably more virucidal than the elemental iodine (Berg, unpublished data).

Figure 8 shows the relationship between time and elemental iodine concentration for the destruction of 99% of coxsackievirus A9 over a three temperature range (Berg, Chang and Harris, 1964). Four other viruses tested under identical conditions were only slightly less resistant. *E. coli, Pseudomonas aeruginosa, Alcaligenes fecalis* and *Staphylococcus aureus* were about equally sensitive to elemental iodine, and depending upon temperature, about 1,100 to 1,700 times more sensitive than coxsackievirus A9 (Berg, unpublished data). The Q_{10} for coxsackievirus A9 was about 4.5, but it was apparently less for *E. coli*.

From Figure 8, it is possible to determine both the times for 99% destruction of the virus over a continuous temperature range at constant I_2 concentrations, or alternately, I_2 concentrations for 99% destruction of the virus over a continuous temperature range at constant times (Berg, 1970).

Ozone

Despite its long widespread use in Europe, there are relatively few data available on the virucidal efficiency of ozone.

Figure 9, however, gives an approximation of what ozone can achieve. Poliovirus 1 was inactivated at a rapid rate in river water with only a few tenths of a mg/l of ozone, apparently at about 25° C

(Coin, Hannoun and Gomella, 1964). Poliovirus 2 and 3 were inactivated more quickly under similar circumstances (Coin, Gamella et al., 1967).

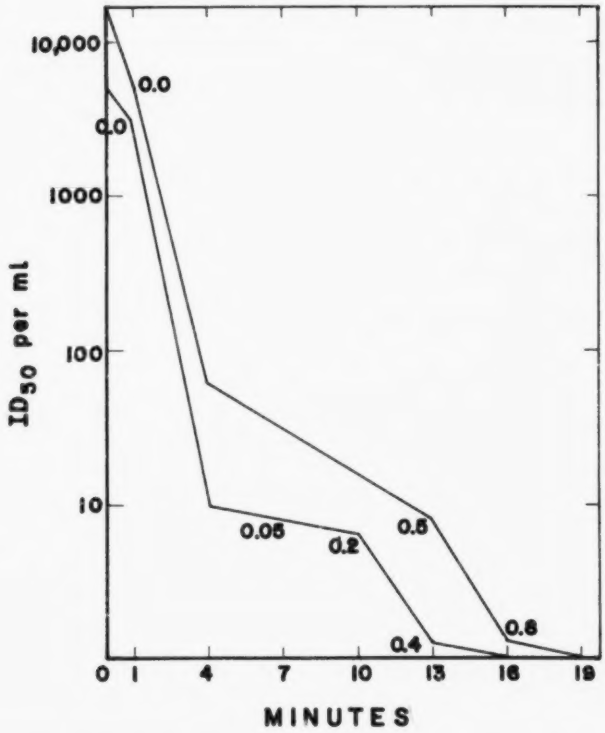

FIGURE 9 Inactivation of poliovirus 1 by ozone. After Coin, et al. (1964)

Ultraviolet Light

Ultraviolet light (UV), at a wavelength of about 250-260 mμ, is strongly bactericidal. Only limited data exist, however, on the virucidal capability of UV light. In distilled water at intensities of 4,000-11,000 μw-seconds per cm, Huff et al. (1965) inactivated at least 99% of polioviruses 1, 2 and 3, coxsackievirus A9, and echovirus 7. Only at lower dosages did small amounts of viruses survive. In the presence of 9 standard units of color, the UV was slightly less efficient. Hill, Hamblet and Banton (1969) evaluated UV, presumedly in the 250-260 mμ range, exposing poliovirus 1 in seawater to 83 $\mu w/cm^2$ of radiation 14 cm from the source. Although the dosage is not completely clear, the rapid virucidal effect of the UV is evident (Fig. 10).

Gamma Radiation

Gamma radiation rapidly destroys viruses. In a menstruum containing salts and serum 0.5 megarads destroyed 90% of poliovirus 3. In distilled water, where quenching of free radicals produced by the radiation did not occur, 90% of the virus was destroyed by less than 0.11 megarads (Sullivan, 1970). Many other viruses tested fell into generally similar resistance patterns. There is little reason to anticipate, however, that economically feasible methods of disinfecting wastewater with gamma radiation will be developed.

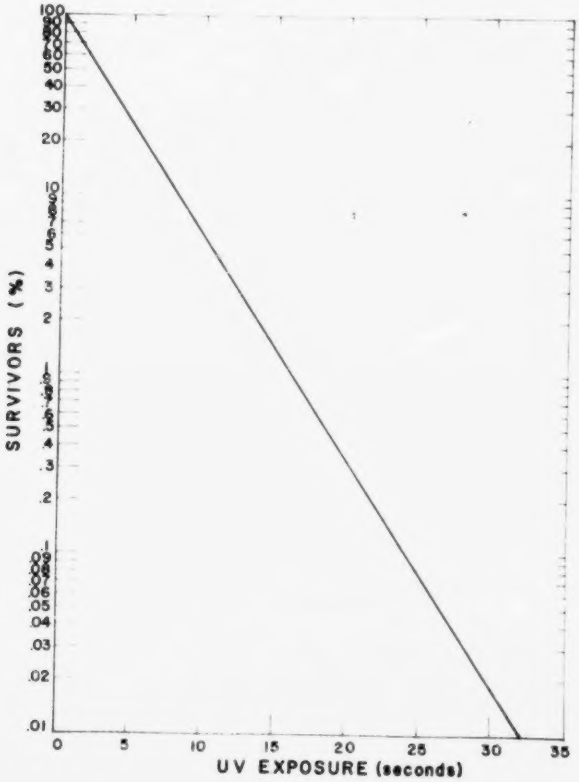

FIGURE 10 Incativation of poliovirus 1 with UV light. After Hill, et al. (1969)

RESEARCH NEEDED

Lotspeich (1969) states, "Design criteria are simply not available - and this includes Scandinavia and Soviet Russia as well as Alaska - to build disposal systems that will function with assurance under the severe climatic conditions of northern latitudes." To those of us interested in disinfection, this indicates that research to solve the problem of effective removal of solids and soluble organic material should have a very high priority if adequate disinfection by chemical methods is to become a practical reality under cold climate conditions.

Information should also be secured to provide satisfactory answers to questions regarding the survival of bacteria and viruses *under natural conditions* in water and bottom sediments under ice. This information is badly needed to determine the effect of an influx of crude and inadequately disinfected wastes on survival patterns under ice cover. Comparable studies of the same receiving waters are needed during open water periods to determine the effect of some of the violent seasonal variations that occur in the Arctic.

It is also well recognized that a significant die-off of coliform and pathogenic bacteria occurs in ice while viruses survive for relatively long periods. What, however, is the effect on these survival relationships when incompletely disinfected effluents are frozen, especially when contact time has been inadequate and some residual disinfectant remains? Research is needed to provide answers to these questions.

The disinfecting potential of lime should be thoroughly investigated. It may be possible to combine successfully the use of lime as a disinfectant with its ability to physically remove microorganisms and a high percentage of solids and oxidizable material. Such research should begin in the laboratory, using raw and/or settled primary sewage at very low temperatures. The effect of increasing pH should be related to the holding time required to yield adequate disinfection under the respective test conditions.

Bench-scale pilot plant studies with lime, or other disinfectants considered subsequently, should be initiated just as quickly as the preliminary research provides adequate guidelines.

Iodine should be investigated in parallel laboratory tests with chlorine using different types of effluents. The inefficient disinfecting properties of chloramines, especially with viruses, combined with the high chlorine demand of many effluents are recognized. These factors create a need for heavy chlorine dosage if adequate disinfection is to be accomplished. The organic demand effects of effluents on iodine are not known. Work is needed to evaluate these effects of effluents on the disinfecting efficiency of iodine. The research outlined should provide a basis for establishing the true comparative efficiency of iodine and chlorine for low temperature disinfection of wastewater.

Preliminary research on the use of ozone should be considered to determine whether ozone demands, especially of the types of effluents to be treated in Alaska, would preclude its use. Some guidelines regarding research with ozone for disinfection of individual household wastes (toilet wastes only) may be available from Smith (1967) who mentions successful treatment of such wastes over a six-month period.

It should be emphasized that disinfection research should get out of the laboratory and into the field for practical evaluation just as quickly as possible. The author is aware of the limited research resources available in most cold climate areas. Nevertheless, wherever possible, it seems desirable to have the laboratory research most closely related to practical application done in the general area where the field evaluation will be made. In this way, personnel most familiar with the technical problems encountered in the laboratory and bench-scale pilot application studies will be available for consultation and assistance in expediting field use.

In conclusion, I should like to point out that all disinfectant research should be planned to provide the best possible basis for determining overall costs for application in various locations. In Alaska, for example, the cost of shipping lime would probably be prohibitive in some inland locations, and ozone would require electrical service. We need to develop a research program to provide a spectrum of proven disinfection processes from which a sanitary engineer can select the one most applicable to a specific waste disinfection problem. Only through such an approach will we be able to satisfy the wastewater disinfection needs of Alaska and similar cold climate areas.

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CRITICAL REVIEW OF PAPERS ON TREATMENT PROCESSES

Karl Wuhrmann

Reviewing eight contributions of excellent and distinguished speakers is a delicate task considering the fact that the most pertinent paper has not been presented at this symposium. I mean the basic text, setting the specific goals and defining the special needs for water pollution control measures in countries with arctic climates. Like in any other geographic region waste treatment has to meet the requirements set by the quality standards which are or have to be imposed on the receiving water bodies by legislation or on the grounds of ecological exigencies. As anywhere, the quality standards of treatment plant effluents must be tailored to the local conditions in rivers and lakes, and the technology of treatment can only be decided upon when these conditions are known. Since this basic concept was lacking, the papers under review necessarily pointed in various directions, according to the personal association of each author to the keyword of "cold climate."

Concerning the first fundamental question, namely "what has to be done?" three objectives have been considered, i.e., the conventional problem of removal of organics, the abatement of eutrophicating effects of wastes and the health hazards involved with sewage. No priorities have been set. All of these objectives might be important, however, at one place or another, and it was justified, therefore, to cover the respective technology to a certain extent.

As to the second fundamental question "how can it be done?" the authors have mostly elaborated on the problem of low temperature effects on process efficiency and design. Process efficiency is undoubtedly an important item in the present context. I should say, however, that it has much less weight in practical water pollution control than such banal items as sludge disposal or purely mechanical problems of operating machinery in an arctic winter. As a microbiologist I am not competent to deal with such technical matters. Having some experience, however, with small treatment plants in the Alps up to altitudes of more than 9,000 ft., I wish to indicate that these technical problems are by far dominating every other question which might arise due to low temperature!

In the group of papers reviewed by Dr. Krenkel it has clearly been shown that the above mentioned objectives one and three, i.e. removal of organics and reduction of health hazards, are most imperative in the majority of Alaskan situations. It was made clear also that, besides a very few large agglomerations where conventional planning of plants may be adequate, mostly small population centers exist, requiring the utmost technical and operational simplicity for any treatment installation.

I take the liberty to comment on the papers under the two headings:

- Basic considerations as to biological processes at low temperatures, and

- Practical observations and experiences with treatment processes.

It is justified to begin with the papers presented by **Eckenfelder** and **Englande** and by **Benedek** and **Farkas**, because the temperature effects on reactions and reaction rates in treatment plants (be they of biological or chemical nature) are fundamental in our context.

It must be emphasized that we have to differentiate between:

- Temperature effects on the performance of the entire treatment plants or individual treatment steps, and

- Temperature effects on single chemical or physico-chemical reactions or reaction chains, including those occurring in organisms.

It would be a fundamental mistake to confound these two groups of temperature effects. In the case of, for instance, a biological treatment plant, the temperature acts preferably on the population dynamics of the entire system. The resultant effect on plant performance is, therefore, not directly related to temperature functions (as for instance the Arrhenius equation) of biochemical reaction rates. It is true, of course, that the population shifts in mixed continuous fermentation systems are caused by temperature effects on growth rates and hence, on biochemical reaction rates. The phenomenologically dominant result, however, is a change in the competition situation of the individual species, leading to a so-called sociological adaptation of the biocenosis (Wuhrmann, 1964). This change within the organism community must not necessarily be associated with a shift in the overall fermentation performance.

In the paper of **Eckenfelder** and **Englande** temperature coefficients of numerous activated sludge laboratory and pilot plant experiments, as well as of lagoons are compiled. The authors, and more extensively **Benedek** and **Farkas**, stress the direct relationship of these observations with the van't Hoff-Arrhenius equation for activation energy. It has to be recognized, however, that these temperature factors are plainly empirical and should not be misused to anticipate the kinetics of specific biochemical reactions. The temperature coefficient of endogenous and substrate respiration of activated sludge has always been a favorite subject for studies and the essential body of **Benedek** and **Farkas'** paper has many predecessors (Sawyer, 1939; Wuhrmann 1955, et al.). θ_{10} values around 2 have been found consistently, from which E_a values of about 11,200 - 12,000 kcal/mol (temp. range around 20° C) may be calculated. This figure is of little significance, however, since we ignore completely the reaction that might be rate limiting within the observable end result of oxygen consumption. It is noticeable that other entirely different reaction chains such as the removal of a substrate from the medium (**Benedek** and **Farkas**) or the kill of bacteria by disinfectants (**Chambers** paper) are also subject to temperature effects with values of θ_{10} in the range of about 1.6 to 2.5 (calculated activation energies around 8,000 - 15,000 kcal/mol). This is a general indication that enzymatically catalized reactions might be involved. It is obvious, however, that just because of this parallelism farther reaching conclusions as to the behavior of complicated systems such as living cells or entire organism associations are not justified.

There is no objection against the determination of temperature coefficients of plant performance. Such values are very useful in practice because they give at least a rough indication as to the magnitude of safety factors to be considered in the dimensioning of biological reactors. It is pertinent, however, to clearly separate 1) reactions in a biological treatment system which are invariably affected by temperature changes according to thermodynamic principles, and 2) the system's behavior as an entity under steady state conditions of practical operation. The classical example for the first group of reactions is sludge respiration with a temperature factor θ_{10} around 2, irrespective of any other environmental conditions or population shifts. It is mandatory, therefore, to consider this temperature factor in the dimensioning of aeration systems. Growth rates as well as substrate resorption rates are subject to similar temperature factors. On a long term basis of continuous operation and at slow temperature shifts, however, these temperature effects will be largely hidden by a sociological adaptation of the sludge which may readily compensate for the change in metabolic rates of the originally dominating species. Most sewage plants demonstrate, therefore, amazingly small shifts of efficiency from summer to winter temperatures. These small differences even disappear more or less completely at low sludge loads (0.1 - 0.2 kg BOD/kg dry solids/day) because substrate supply and not temperature is then acting as the rate limiting factor for sludge metabolism. I think there is no need for **Eckenfelder's** and **Englande's** hypothesis, assuming sludge flocculation is an essential parameter for the magnitude of temperature effects on the overall performance of conventional activated sludge systems. This hypothesis is even contradicted by earlier experimental evidence such as Sawyer's observations (1955) and the reviewer's own results (Wuhrmann, 1964). The idea of temperature-independent rate limitations by too small a substrate supply is further confirmed by the skim milk experiments in the paper of **Koyama et al.** and by the laboratory experiments described by **Clark et al.**

The essential practical conclusions are therefore:

1. Compensation for adverse effects of low temperature in activated sludge systems is readily possible by low sludge loads. With domestic sewage the limit is in the order of magnitude of 0.1 to 0.2 kg BOD/kg MLSS/day. As was shown by the reviewer (Wuhrmann, 1964) and has been confirmed with the skillful experiments of **Clark et al.**, dimensioning has to consider the *lowest* temperatures occurring.

2. Aeration rates have to be evaluated on the basis of the *highest* temperatures to be expected in the system. Exigencies for low temperature will then automatically be satisfied.

3. Although lagooning might be a relatively cheaper investment than other systems, it is doubtful it will serve the needs, emphasizing the fact that the highest treatment efficiency is required in wintertime, i.e. at the lowest temperatures (see introduction of **Gordon's** paper).

The above conclusions lead to some technical consequences regarding sewage and plant construction. From the operational point of view one of the main problems - especially with small plants - is ice formation. Heat conservation is imperative, therefore, and requests a very compact and condensed plant layout. Heavily exposed plants in the Alps of Switzerland are enclosed for protection against snow and wind and for easier maintenance. The largest heat loss occurs in the aeration basin due to evaporation and intensive exposure of the water to the atmosphere. As has already been mentioned by **Pick** and others, surface aerators should, therefore, not be used. Very much can be done in favor of safe plant operation with an adequate sewage system as has neatly been shown in the paper of **Koyama et al.** He demonstrated how thawing may interfere with treatment efficiency by increasing the hydraulic plant load and simultaneously decreasing the sewage concentration and temperature. All three factors may work together and produce an acute danger of sludge washout. The answer to this problem evidently is separate sewering, a concept which is already adopted in the United States but still meets resistance in other places.

Not many words have been devoted at this conference to the delicate topic of sludge treatment and disposal, the paper of **Balmér** excepted. The problem is highly critical with small installations where no machinery for sludge processing can be afforded. It is a favorable coincidence, however, that small plants should be designed as very low loaded complete oxidation units, for reasons already discussed. This concept automatically involves a minimum of excess sludge production and a sludge quality which is inoffensive, creates no odor problems and dries rapidly (especially after freezing). Under these conditions the conventional drying bed is probably the optimum solution for sludge disposal.

The question of eutrophication by plant effluents has been discussed in the papers of **Rosendahl** and **Balmér** in regard to phosphorus removal from sewage. This is of course an immediate problem all over the world. Irrespective of the obvious question of whether nutrient removal from wastes is of top priority in the Alaskan situation, the process described by **Balmér** raises an interesting point worth mentioning: chemical processes such as precipitation or adsorption etc. have much lower temperature coefficients (diffusion being mostly the rate-limiting process) than enzymatically catalized reactions. Dimensions and operations of chemical purification plants are much less affected, therefore, by temperature shifts than are biological units. This represents a considerable advantage. In view of the high degree of purification required by the winter conditions in arctic rivers, it is questionable, however, whether exclusively chemical processes as described by **Balmér**, are sufficiently effective in regard to removal of dissolved organic compounds. It is also common experience that handling of excess sludge from precipitation units can be a difficult task. I am of the opinion, therefore, that the application of chemical precipitation or flocculation processes under critical operation conditions (small plants, low temperature, unskilled personnel, etc.) should be considered with caution.

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